Biophysical Indicators of Sustainability for Climate Change Mitigation and Adaptation

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Abstract

The creation of the Sustainable Development Goals (SDGs) offers a great opportunity to look at sustainability from a different perspective. Identifying specific indicators, metrics, and spatial and temporal boundaries help improve definitions of sustainability to work toward more sustainable systems. A hierarchical framework for evaluating sustainability puts ecological or environmental components first, since all social and economic factors depend on a life-giving Earth system. The overall research question in this dissertation is: How do changes in land use or land cover relate to climate change and biophysical sustainability in complex systems? Three individual case studies are conducted to address this question along with aspects of proposed SDGs. From drought in the US Great Plains, increased precipitation frequency in northwestern Brazil, and an increased urban heat island in a mega-city in southeastern Brazil, the case studies demonstrate some of the ways in which climate change and sustainability intersect. By exploring different biophysical and ecological indicators, explicitly accounting for spatial and temporal scale, and utilizing cross-disciplinary methodologies, the studies quantify environmental sustainability and, not surprisingly, find an overall move away from biophysical sustainability. In so doing, they also reveal pathways that can be followed to move agricultural, urban or forest systems back toward greater sustainability. These pathways can contribute not only to increased sustainability but also to climate change mitigation and adaptation at local to regional scales resulting in win-win situations. Additionally, the findings of each study contribute to four of the 17 proposed SDGs, offer inputs for policy considerations, and build on theory and conceptual frameworks for future applications of sustainability science.

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Chapter 1

Introduction

1.1 Sustainable Development Goals

At the Rio+20 United Nations Conference on Sustainable Development in 2012, nations decided to create the Sustainable Development Goals (SDGs) as a follow-up to the Millennium Development Goals (MDGs) which expire at the end of 2015. An Open Working Group (OWG), representing 70 countries and endorsed by the UN General Assembly, proposed a broad list of 17 goals and 169 targets. The Report of the Open Working Group of the General Assembly on Sustainable Development Goals (UN A/68/970) was released for two public consultations where feedback was provided by organizations, scholars, and citizens from around the world.

The UN General Assembly is set to discuss the proposed SDGs in summer 2015 in order to announce the final goals and targets in September 2015. The SDGs are a voluntary agreement, not a binding treaty, but they are meant to guide sustainable development around the world for the next 15 years, through 2030. However, after a rigorous analysis of the proposed goals and targets, the International Council for Science (ICSU) and the International Social Science Council (ISSC) reported that only 29% of the 169 targets were well-defined with specific endpoints, time frames, and possibilities for accurate, scientific measurement (Stokstad, 2015). Numerous scholars have criticized aspects of the proposed SDGs, especially in relation to global and environmental ethics (Brandi, 2015; Fukuda-Parr & McNeill, 2015; Pogge & Sengupta, 2015; Pongiglione, 2015; Watene & Yap, 2015), making suggestions for improvements in the last few months before they are adopted.

From implementation to indicators and accountability, the current SDGs seem overloaded with details in spite of ignoring key issues of ethics, diversity, and planetary boundaries. While Pongiglione (2015) argues for a priority structure of the SDGs to ease implementation, Fukuda-Parr & McNeill (2015) call for a global justice approach to accountability, recognizing the limitations of accountability mechanisms in certain nations and calling on international organizations and decision-making bodies. According to Watene & Yap (2015), Indigenous communities are disappointed that the proposed SDGs do not address the importance of culture as an aspect of development. Since 24% of global lands are designated Indigenous territories, where 80% of global biodiversity can be found, Indigenous knowledge or the loss thereof should be recognized as an indicator of sustainability (Watene & Yap, 2015).

Brandi (2015) points out that the proposed SDGs do not contain the goal to safeguard our Earth's life-support system on which all future global development depends. Somewhere along the way between the first and second public consultation of the SDGs, the initially proposed Goal 2 to "Achieve Development within Planetary Boundaries" was changed to "Promote Economic Growth and Decent Jobs Within Planetary Boundaries," and then the phrase "planetary boundaries" was dropped completely from all goals (review of documents at http://unsdsn.org/resources/publications /indicators/). Considering that planetary boundaries for climate, biodiversity and human interference in the biogeochemical nitrogen cycle have already been crossed (Rockström et al., 2009) and despite calls from renowned global scholars (Griggs et al., 2013; Rockström & Sachs, 2013), the OWG seems preoccupied with an alliance of Organization for Economic Co-operation and Development (OECD) countries and powers that fear limitations on development. To which Brandi (2015, pg. 2) argues that respecting planetary boundaries (or "guard rails") is "an indispensable precondition for poverty eradication and development, and the guard rails do not restrict the implementation of the development goals that state that all people should be given access to food, safe water and sustainable energy."

When discussing how climate change relates to sustainable development goals, Craig (2014, pg. 2) notes: "At some point, in other words, a society's dependence on a failing or radically changing ecosystem drastically retards, even reverses, economic and social development." The IPCC's Fifth Assessment report further corroborates the impact that climate change will have on the proposed SDGs: "[a]dded to other stresses such as poverty, inequality, or diseases, the effects of climate change will make sustainable development objectives such as food and livelihood security, poverty reduction, health, and access to clean water more difficult to achieve for many locations, systems, and affected populations" (Field et al., 2014).

While there is plenty of evidence linking sustainable development and climate change, quantitative studies that attempt to address both simultaneously are few. Part of this disparity results from the ever evolving definition of what we mean by sustainability. Even in the SDGs, Camacho (2015) recognizes that the definition of "sustainable" in sustainable development is still not clear for certain situations.

1.2 Defining Sustainability

The most widely known definition of sustainability is from the Brundtland Report on sustainable development: "development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987)." Over and over, the literature reveals consensus on the ambiguity of such a broad definition. Profound differences in disciplinary interpretations exist, and with them difficulties in pinpointing the right composite index or multivariate indicator to assess sustainability at varying spatial and temporal scales (Böhringer & Jochem, 2007; Brown et al., 1987; Mayer, 2008; Mori & Christodoulou, 2012; Singh et al., 2009). For example, Mori & Christodoulou (2012) have conducted a thorough review of the pros and cons of over 15 sustainability indices. In that review, the authors identify two common elements of all sustainability definitions: 1) the "triple bottom line" of economic prosperity, social justice, and environmental quality (Elkington, 1998), and 2) intergenerational equity. They conclude that none of the indices reviewed cover both the triple bottom line and external leakage impacts for a common evaluation of sustainability in cities around the world.

Anthropogenic climate change is often considered as a biophysical indicator that we are moving away from sustainability on a global scale. Given this global problem, how useful is it for us to evaluate sustainability on local (city) to regional (nation-state or national) scales? From a policy perspective, there are advantages in constraining assessments to political boundaries. However, leakage impacts make it difficult to quantify which locale is contributing more or less to climate change mitigation and sustainability of the planet as a whole. An additional difficulty is the need to account for and separate out the local to regional impacts of climate change, which may contribute to or take away from a region's sustainability. Synergies between climate change mitigation and sustainable agricultural practices, for example, can provide for local adaptation (Wall & Smit, 2005). But, what happens when global climate change results in drought or flood events that trump local sustainability efforts?

For the first and third case studies in this dissertation, sustainability in agroecosystems is defined as: the maintenance of ecological systems for agricultural productivity on behalf of present and future generations despite constraints and disturbances related to socio-economic pressures. Since the early development of agroecology in the 1920s, there have been continued efforts in the application of ecological science to the design and management of agroecosystems for purposes of sustainability (Altieri, 1987; Conway, 1985; Wezel et al., 2009). In light of anthropogenic climate change, the Food and Agriculture Organization (FAO) of the United Nations (UN) has recently introduced another term and framework for the management of sustainable agroecosystems and livelihoods. Climate-smart agriculture is defined as an agricultural system that "sustainably increases productivity, resilience (adaptation), reduces/removes greenhouse gases (mitigation) while enhancing the achievement of national food security and development goals" (Lipper et al., 2010). Achieving climate-smart agroecosystems requires a high level of diversity in land cover and species composition, a focus on land-use interactions, and integrated landscape management at a landscape scale (Scherr et al., 2012).

For the second case study, urban sustainability is defined in terms of more sustainable urban forms or: biophysical forms that maintain connectivity, complexity (diversity or mixed land use), greening, and scale- or location-specific levels of density (compactness or aggregation) that mitigate the urban heat island (UHI) and allow for adaptation to climate change impacts. Combining concepts from transect planning (Duany & Talen, 2007), biophilic urbanism (Beatley & Newman, 2013), and design themes from neotraditional development, urban containment, compact cities, and eco-cities (Jabareen, 2006), over time there should be an increase in connectivity, diversity, and greening for improved urban sustainability. However, when it comes to density (compactness or aggregation), Burton et al. (2013) conclude that we should look instead for various urban forms that are appropriate depending on location and scale.

Mayer (2008) discusses the importance of the identification of spatial and temporal boundaries in the assessment of sustainability in human-environment systems. Ecoregions may be a more appropriate scale for evaluating environmental impacts, which also benefit from the availability of a longer time series of data. On the other hand, political boundaries may better allow for the evaluation of economic and social tradeoffs, which often interact with policies on much shorter time scales. The decoupling of policy actions from sustainability targets, like reducing anthropogenic climate change, is often attributed to the discrepancies between spatial and temporal scales and is the cause of an ever-widening sustainability gap (Fischer et al., 2007a).

To close the gap between meaningful sustainability targets and our present trajectory, Fischer et al. (2007a) prefer to see the ecological, social, and economic components of sustainability as a nested hierarchy rather than the parallel pillars of the triple bottom line. Ecological or biophysical considerations are first in this hierarchy, for all just societies (second) and economic prosperity (third) must rely on our life-giving Earth system. This nested, hierarchical conceptualization initially garnered attention when the UN first proposed Sustainable Development Goals, and along with it a new definition of sustainable development was considered: "development that meets the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generations depends" (Griggs et al., 2013).

Although this definition seems to be less important for current SDGs, it is this overall definition of sustainable development that the case studies in this dissertation follow. Safeguarding Earth's life support systems requires mitigation of anthropogenic climate change, and the welfare of current and future generations depends on climate change adaptation. Therefore, the purpose of this dissertation research is to assess the sustainability of varying ecosystems, from agricultural to urban, by exploring synergies and tradeoffs between biophysical metrics or indicators of sustainability and climate variables.

1.3 Research Questions and Objectives

The overall research question in this dissertation is: **How do changes in land use or land cover relate to climate change and biophysical sustainability in complex systems?** Three individual case studies address this question along with some aspects of four of the 17 proposed Sustainable Development Goals. These goals include:

- Goal 2: End hunger, achieve food security and improved nutrition, and promote sustainable agriculture
- Goal 11: Make cities and human settlements inclusive, safe, resilient and sustainable
- Goal 13: Take urgent action to combat climate change and its impacts
- Goal 15: Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss

By employing a multidisciplinary methodology and eddy-covariance (EC) data, the first research question in Chapter 2 tackles aspects of Goals 2, 13, and 15: **How does thermodynamic entropy production as a measure of sustainability in agroecosystems relate to water and carbon cycling in the Central US?** In this case study, agroecosystem sustainability was assessed using thermodynamic entropy production (σ) and its relation to water, energy and carbon cycling in croplands and grasslands of the Central US. From 2002 to 2012, the biophysical metric of σ was compared across AmeriFlux sites (Baldocchi et al., 2001), each with site-specific land management practices of irrigation, crop rotation, and tillage, to provide an improved understanding of spatial and temporal variability, agroecosystem sustainability and climate impacts.

In Chapter 3, Goals 11 and 13 are addressed by asking the question: **How can MODIS satellite products be employed for long-term monitoring of biophysical metrics that relate to more sustainable urban forms at the 1-km scale?** To test metrics for rapid identification and global evaluation of more sustainable urban forms, the configuration of the São Paulo Metropolitan Region (SPMR) in Brazil is examined using satellite remote sensing data and landscape metrics. The purpose of this study is to uncover a relatively simple and rapid way, given any urban classification system, to evaluate urban sustainability at the global scale.

The case study in Chapter 4 again focuses on aspects of Goals 2, 13, and 15, and shows how Indigenous knowledge can be an indicator for more than just biodiversity loss: **What are appropriate spatial and temporal scales for assessing climate change impacts and sustainability in the Tiquié River basin?** This research attempts to deliver useful climate services for sustaining Indigenous livelihoods by leveraging results from participatory research of the Tukano ecological calendar to identify temporal and spatial scales for evaluating climate change and sustainability. Given the importance of agroecosystems and fisheries production in the Tiquié basin, identifying timescales that relate to the slash-and-burn practice and fish reproduction and migration cycles is key to evaluating sustainability in this region. Indicators of Indigenous ecological knowledge, precipitation, and river levels are combined with cross-disciplinary methodologies to examine temporal and spatial variability.

Taking an integrative approach, the research in this dissertation is broad and exploratory with a goal of simplicity and a focus on biophysical indicators at local to regional scales. To achieve sustainability, Fischer et al. (2007a) recommends a three-pronged approach: 1) identify meaningful biophysical targets, 2) apply policy tools to promote sustainable actions, and 3) identify key foundational or ethical values that link short-term policy actions with long-term sustainability targets. While the proposed research will not include an ethical component, it is understood that sustainability in any system at any spatial or temporal scale cannot be attained without consideration of key foundational common truths for humanity. The First Principles of Agroecology: 1) life has purpose, 2) all life is interconnected, and 3) life is good, serve as an example of a common sense foundation on which scientific exploration of environmental sustainability can be built (Ikerd, 2010). These guiding Principles will be kept in mind, however, they will not be incorporated into methodology or interpretation of results at this time.

Combining strategies from Mayer (2008) and Fischer et al. (2007a), it follows that research on environmental sustainability can benefit from: 1) definition of spatial scale, 2) definition of temporal scale, 3) identification of biophysical indicators/targets that have value to society, and 4) definition of an adaptive policy tool to work toward sustainability. Because the purpose of this dissertation as a whole is not a comparison between regions, different biophysical indicators will be selected and appropriate assessment tools employed to perform each analysis. Since environmental sustainability in light of climate science calls for optimization of location-specific practices (Selvaraju et al., 2011), the process of selecting meaningful indicators begins with assumptions regarding the usefulness of metrics to get at the most pertinent stressors and limiting factors identified in each region. Despite stark differences in geographic areas and research methods, it is hoped that common synergies and tradeoffs will emerge and be useful for future understanding of environmental sustainability on local to regional levels given global climate change.

1.4 Organization of the Dissertation

Three different case studies are explored in this dissertation, found in Chapters 2, 3 and 4. Each case study is formatted as a journal article with Introduction, Methods, Results, Discussion and Conclusion sections. The Introduction of each Chapter includes background, literature review, and statement of research problems, questions and objectives relevant to that study. The Methods section describes the data and methodology employed in each analysis, while the Results, Discussion and Conclusion sections outline the findings, contributions, sources of error, and directions for future research.

In Chapter 5, a summary of how each Chapter contributes to an overall improved understanding of environmental sustainability in relation to climate change and the Sustainable Development Goals is provided. This discussion includes policy implications and suggestions for future research.

Chapter 2

A Thermodynamic Approach for Assessing Agroecosystem Sustainability

Ferdouz V Cochran, Nathaniel A Brunsell, Andrew E Suyker

Abstract

By revisiting theoretical concepts in biogeography and the importance of thermodynamic laws in biosphere-atmosphere interactions, biophysical sustainability in agricultural systems can be better defined. In this case study, we employed a multidisciplinary methodology for examining agroecosystem sustainability by using eddy-covariance data to compute thermodynamic entropy production (σ) and relate it to water, energy and carbon cycling in croplands and grasslands of the Central US. From 2002 to 2012, the biophysical metric of σ was compared across AmeriFlux sites, each with site-specific land management practices of irrigation, crop rotation, and tillage, to provide an improved understanding of spatial and temporal variability and climate impacts. Results show that σ is most correlated with net ecosystem exchange (NEE) of carbon. When cropland and grassland sites are close to being carbon neutral, σ values range from 0.51 to 1.0 $WK^{-1}m^{-2}$ for grasslands, 0.81 to 1.0 $WK^{-1}m^{-2}$ for rainfed croplands, and 0.81 to 1.1 $WK^{-1}m^{-2}$ for irrigated croplands. Irrigated maize stressed by hydrologic and high temperature anomalies associated with the 2012 drought exhibit the greatest increase in σ, indicating decreased sustainability compared to rainfed croplands and grasslands. These results suggest that maximizing carbon uptake with irrigation and fertilizer use tends to move agroecosystems away from thermodynamic equilibrium and sustainability, which has implications for food security and greenhouse gas (GHG) mitigation for climate-smart agriculture. The underlying theoretical concepts, multidisciplinary methodology, and use of eddy-covariance (EC) data in this study contribute to a unique understanding of biogeography and human-environment interactions in agricultural systems creating opportunities for future work.

2.1 Introduction

The role of thermodynamics is inherent in the field of physical geography, exemplified through abiotic-biotic or spatial interactions. Discussions concerning non-equilibrium dynamics and irreversible processes are seen in biogeography, climatology, geomorphology, and landscape ecology (Brunsell et al., 2011; Holdaway et al., 2010; Perry, 2002; Phillips, 1999, 2008; Smith, 2005; Steinborn & Svirezhev, 2000; Svirezhev, 2000) and are applicable to all natural and anthropogenic landscapes, including agricultural systems on varying temporal and spatial scales. In this study, we tie together thermodynamic laws, theories of complex ecosystem dynamics, and resilience theory to explore biophysical aspects of sustainability in agroecosystems. We propose that higher thermodynamic entropy production (σ) indicates higher stress and a move away from thermodynamic equilibrium and adaptive potential. The σ metric is used to assess geographic variability in sustainability among different land management strategies in the Central US.

2.1.1 Thermodynamics and ecosystem evolution

Thermodynamic entropy has been explored for evaluating sustainability in various disciplines including industrial ecology, resource economics, and mechanical engineering (Gutowski et al., 2009; Hermanowicz, 2007; Krotscheck, 1997; Liao et al., 2012; McMahon & Mrozek, 1997). Different frameworks for evaluating systems and subsystems have been proposed, but complexities in applying these frameworks remain. The major difficulty in conducting and interpreting studies, manifesting in numerous debates and misunderstandings across disciplines, and even between Nobel Laureates (Gnaiger, 1994), relates to the terminology used in describing the systems studied and in identifying spatial and temporal boundaries. It is through identification of these system boundaries and improved understanding of abiotic-biotic or spatial interactions that the field of physical geography is ideally equipped to contribute to the future of sustainability science.

Based on the laws of thermodynamics, the physical boundaries for interactions between the Universe, the Earth and its ecosystems can be outlined as follows: the Universe is an isolated system that does not have inputs and outputs of energy; the Earth is a closed system that exchanges energy with the Universe but, for all purposes, not matter; and ecosystems are open systems exchanging both energy and mass within the Earth system. The second law of thermodynamics helps us understand ecosystems as open systems with dissipative structures far from thermodynamic equilibrium that evolve to maintain a high level of local organization resulting in a production of entropy (Schneider & Kay, 1994). Schneider & Kay (1994) argue that ecosystems, and the species that thrive within them, develop an increasing ability to degrade incoming solar radiation, which increases the total dissipation of heat from that ecosystem. This part of energy that is no longer available for work is also known as thermodynamic entropy.

Thermodynamic entropy results from non-equilibrium thermodynamic processes occurring in a system. It is produced by fluxes of heat, matter, and momentum and related to gradients of temperature, pressure, concentration, etc., which maintain systems away from equilibrium. Within the closed Earth system, solar radiation creates a large influx of energy and a gradient that open ecosystems will strive to reduce through all available chemical and physical processes. Ecosystems that are less stressed tend to exhibit a greater ability to degrade solar energy and reduce gradients than stressed ecosystems (Schneider & Kay, 1994).

From an ecosystem development perspective, entropy production may be at a maximum during three developmental stages due to: 1) early successional growth and rapid colonization by fast growing species, 2) sustained production during maturity of longer-growing species, and 3) extended maturity or delay of retrogression by stress-tolerant species (Holdaway et al., 2010). During the growth stage, species with rapid population growth may have a higher initial entropy production than slower growing species. The temporal and spatial scales of biogeochemical and biogeophysical processes involved add to the complexity of evaluating the maximum entropy production (MEP) hypothesis, which suggests that the maximum rate of entropy production in a system occurs when the influence of vegetation productivity on land surface albedo and the effect of solar radiation absorption on evapotranspiration result in maximizing the energy flux or dissipation from the system (Kleidon, 2009).

Changes in total entropy of a system result from all the entropy production and transfer processes associated with that system. Brunsell et al. (2011) quantified the thermodynamic entropy budget of the land surface, calculating both entropy production and transfer, and found that a higher vegetative fraction results in increased entropy production and a decreased rate of change in total entropy. It is important to note that the decrease in the total entropy budget is related to the decrease of the Bowen ratio (the ratio of sensible to latent heat fluxes) or the increase of entropy transfer associated with latent heat flux during the daytime and sensible and soil heat fluxes during the nighttime. Brunsell et al. (2011) also applied their methodology to data from three eddy covariance (EC) flux towers in northeastern Kansas with different land cover types and land use management practices. Results suggest that the overall ecosystem entropy is related most to net radiation at the land surface, and that entropy production is driven by land cover and land management.

Disturbance plays an integral role in the structure of ecosystems at multiple scales creating a landscape mosaic where there are interactions between the heterogeneous surface properties or patchiness of the landscape and ecological processes (Perry, 2002). Phillips (1999, 2008) outlines the thermodynamic principles behind numerous theories of ecosystem structure, function, and development, where evolution itself can be thought of as a fundamental irreversible process. Phillips (2008, pg. 56) states that "perhaps the most robust theory of evolution at the ecosystem and biospheric scale" is one presented by biogeographer Charles H. Smith, an expert on Alfred Russell Wallace.

Smith (2005) interprets Wallace's view of evolution as one based on spatial interactions between species, versus adaptations within species. In Wallace's perspective, Smith argues, there is no process of adaptation only the result of being adapted. In this paradigm, evolution of spatial interactions occurs between abiotic and biotic components in a systems framework, which contains both negative and positive feedback loops (Smith, 1986, 2005). "Adaptive structures" are part of a negative feedback loop where "deviation-countering" processes are maintained at the organismal level through biogeochemical cycling and dissipation of energy, hence the production of entropy. "Adaptive potential" and the selection of traits at the ecosystem level entails a positive feedback loop where "deviation-amplifying" processes occur and spatial interactions evolve. The divergence associated with evolutionary change and adaptive potential is a return toward instead of a move away from thermodynamic equilibrium. Using theoretical models, Kostitzin (1934) also described evolution as a series of unlikely events opposing the increase of thermodynamic entropy. Consequently, the energy and directionality that exist in working against change in the negative feedback loop may increase thermodynamic entropy production, and the randomness associated with genetic mutations and probabilistic spatial interactions related to evolutionary change in the positive feedback loop will correspond with lower entropy production (Phillips, 2008; Smith, 1986, 2005).

Organisms and ecosystems may also evolve in directions of lower stress (Smith, 1986, 1989, 2005). If we consider stress as spatial gradients related to the negative feedback part of the evolution framework described above, more intense gradients will require more work to maintain "deviation-countering" processes resulting in higher entropy production. Thus, thermodynamic entropy production can be an indicator of higher stress where the system is maintained in the negative feedback loop moving away from thermodynamic equilibrium versus entering the positive feedback loop where "deviation-amplifying" processes bring a system back toward equilibrium. It is important to recognize that living systems do not actually reach thermodynamic equilibrium because they are continuously exchanging energy and matter with their environments as open systems, as previously explained. However, it is the move back toward thermodynamic equilibrium that is crucial to Smith and Wallace's evolutionary theory and the result of adaptation.

Anthropogenic inputs of fertilizer and irrigation in agricultural systems can force organisms to remain in the negative feedback loop, creating larger gradients with their surroundings corresponding to higher stress and entropy production. Anthropogenic impacts on agroecosystems have previously been quantified with thermodynamic entropy metrics using data on agricultural inputs and outputs related to tillage, fertilization, pesticide use, harvest, etc. (Steinborn & Svirezhev, 2000; Svirezhev, 2000; Patzek, 2008). Energy inputs in agricultural systems can lead to an overproduction of entropy, and the greater the overproduction, the less sustainable a system is said to be (Steinborn & Svirezhev, 2000; Patzek, 2008). Svirezhev (2000) developed a methodology to estimate an "entropy fee" that humans pay for intensive agricultural production, where sustainability is ultimately achieved when the "entropy fee" equals zero. Steinborn & Svirezhev (2000) calculated the entropy balance for different crops over almost a decade and found that the "entropy fee" (also referred to as the overproduction of entropy or environmental degradation) can be diminished through a reduction of anthropogenic inputs, especially fertilizers. Patzek (2008) revised their methodology to examine the sustainability of US maize production and concluded that large monocultures are not sustainable.

2.1.2 Biophysical sustainability

With compounding pressures of a growing global population, increased food demand, and changing climatic conditions there is an urgent need to understand the geographic variability of sustainability in agricultural systems. Sustainability in agroecosystems has been defined as the maintenance of productivity over time despite socio-economic pressures and ecological constraints and disturbances (Altieri, 1987; Conway, 1985; Wezel et al., 2009). Recently, the Food and Agriculture Organization (FAO) of the United Nations (UN) has introduced a framework for the management of sustainable agroecosystems and livelihoods given anthropogenic climate change. Climate-smart agriculture is defined as an agricultural system that sustainably increases agricultural productivity and income, builds adaptation and resilience to climate change, and contributes to mitigation or removal of greenhouse gases (Lipper et al., 2010). Achieving climate-smart agroecosystems requires a high level of diversity in land cover and species composition, a focus on land-use interactions, and integrated landscape management at a landscape scale (Scherr et al., 2012).

While many still view the triple bottom line for sustainability as parallel pillars that must be addressed equally, Fischer et al. (2007a) prefer to see the ecological, social, and economic components of sustainability as a nested hierarchy. In this approach, ecological or biophysical considerations are primary, for all just societies (secondary) and economic prosperity (tertiary) must rely first on our Earth system. This framework is appropriate when examining sustainability on a global scale. For example, by evaluating our Earth's energy budget, scientists have concluded that anthropogenic climate change is a biophysical indicator that our Earth system is moving away from sustainability. In addition, Kleidon (2009) argues that it is useful to consider Earth's entropy budget, since non-equilibrium thermodynamic entropy production is a measure of dissipative processes and may provide an estimate of climate sensitivity.

At anything less than a global scale, the challenge in evaluating sustainability becomes identifying appropriate spatial and temporal boundaries for the assessment of human-environment systems (Mayer, 2008). The identification of system boundaries is essential for acknowledging leakage effects where resource inflows may or may not be sustainable (Mori & Christodoulou, 2012), and legacy effects where human-environment couplings can vary from decades to centuries (Liu et al., 2007). In addition, the usefulness of sustainability metrics remain uncertain when they do not get at the underlying mechanisms or processes that move systems toward or away from sustainability.

In resilience theory, building and maintaining resilience through the process of the adaptive cycle is an essential part of working toward long-term sustainability. The ability to continue adapting and changing yet remain within critical thresholds makes a system more resilient (Walker & Salt, 2012). Both a system's resilience and its efficient use of resources/energy contribute to its sustainability (Patzek, 2008; Walker & Salt, 2012).

The adaptive structures in the negative feedback loop and the adaptive potential in the positive feedback loop from Wallace's evolutionary theory can be combined with the adaptive cycle of resilience theory to relate thermodynamic entropy production to biophysical sustainability. Evolutionary divergence or adaptive potential in the positive feedback loop is characterized by a return toward thermodynamic equilibrium. Hence, a move away from thermodynamic equilibrium with higher entropy production indicates less adaptive potential in an open ecosystem, which would indicate less resilience and sustainability. At the same time, higher entropy production signals the breakdown of more intense or persistent gradients corresponding to higher stress, which may indicate inefficient energy or resource use also indicating less sustainability.

2.1.3 Case Study

Given the need for a better understanding of the geographic variability of sustainability utilizing different land use management strategies, the increasing availability of EC tower data through AmeriFlux and other regional networks allows for empirical studies. The micrometeorological data from these networks enables computation of thermodynamic entropy production in ecosystems across the US and other parts of the world. Without the additional calculations related to anthropogenic inputs of energy (Svirezhev, 2000), the methodology provided by Brunsell et al. (2011) may facilitate the use of thermodynamic entropy for assessing sustainability. Therefore, this case study examines agroecosystems from the perspective of thermodynamics and evaluates the use of entropy production as a biophysical metric for agroecosystem sustainability across the geographic region of the Central US.

The overarching question of this case study is: How does thermodynamic entropy production as a measure of sustainability in agroecosystems relate to water and carbon cycling in the Central US? It is hypothesized that: Agroecosystems that are stressed by hydrologic anomalies will exhibit increased thermodynamic entropy production indicating decreased sustainability. To test this hypothesis, thermodynamic entropy production is computed using net radiation and individual radiation terms from AmeriFlux site data in select croplands and grasslands in the Central US. Variability of thermodynamic entropy production is then compared to changes in soil water content, energy partitioning indicated by the Bowen ratio, and net ecosystem exchange of carbon. While the need for a "standard environment" as a reference level for the evaluation of entropy production and sustainability is debatable (Hermanowicz, 2007; Krotscheck, 1997), we have chosen to include the Kansas grasslands as baseline ecosystems against which to compare entropy production in the Nebraska and Oklahoma agroecosystems.

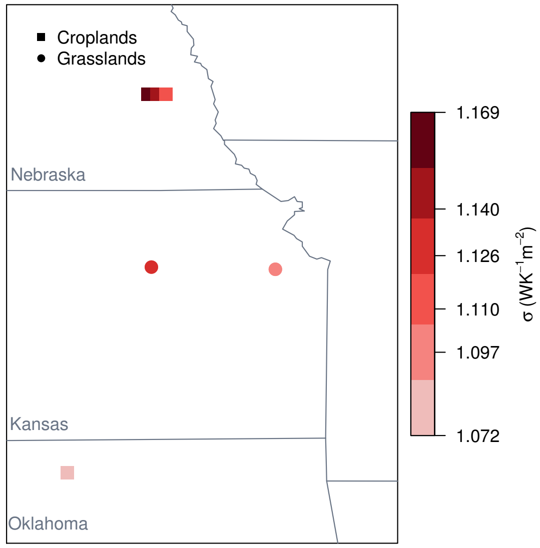

Figure 2.1: AmeriFlux tower locations (not to scale) for grassland and cropland sites in Kansas, Nebraska, and Oklahoma. Colorbar indicates mean hourly σ from monthly daytime (9am-4pm) averages across all years of available data at each site.

2.2 Methods

2.2.1 AmeriFlux cropland and grassland sites

Thermodynamic entropy production was evaluated for six AmeriFlux sites in Kansas, Nebraska, and Oklahoma (Figure 2.1) with differences in land cover and land management practices. All sites are within the humid continental Köppen climate classification, though the Oklahoma site is sometimes categorized as humid subtropical. Table 2.1 provides a summary of site characteristics with average daytime hourly values for incoming solar radiation (Q_S, Wm^{-2}), entropy production (σ, WK^{-1}m^{-2}), soil water content (SWC, %), the Bowen ratio (β), and net ecosystem exchange of carbon (NEE, μmol C m^{-2}s^{-1}). Further description and land use histories of Kansas grassland sites are given in Cochran et al. (2013). Details on the Nebraska cropland sites, which are within 1.6 km of each other, are outlined in Verma et al. (2005), and information on the Oklahoma cropland site can be found in Fischer et al. (2007b).

The data obtained at these sites and used in the analysis are from micrometeorological EC

Table 2.1: AmeriFlux EC tower site characteristics, including site years available, International Geosphere-Biosphere Programme (IGBP) class, dominant vegetation, management, mean annual temperature (T), mean annual precipitation (PPT), and average hourly values between 9am-4pm across all available site years of incoming solar radiation (Q_S), entropy production (σ), soil water content (SWC), the Bowen ratio (β), and net ecosystem exchange (NEE).

Sites	**ARM**	**KFS**	**KON**	**NE1**	**NE2**	**NE3**
Years	2003-2012	2008-2012	2007-2012	2002-2012	2002-2012	2002-2012
IGBP Class	Cropland	Grassland	Grassland	Cropland	Cropland	Cropland
Dom Veg	Winter Wheat	C_3 Grasses	C_4 Grasses	Maize	Maize/Soy Rot*	Maize/Soy Rot
Management	Rainfed	Rainfed	Rainfed	Irrigated	Irrigated	Rainfed
	Till	5yr Burn/Mow	Annual Burn	Till/Cons Plow	Till/Cons Plow	No Till
Mean Annual PPT	600mm	937mm	835mm	839mm	861mm	697mm
Mean Annual T	15°C	13°C	13°C	11°C	10°C	10°C
Q_S **(**$\mathbf{Wm^{-2}}$**)**	492.213	448.182	474.511	456.094	462.057	452.909
σ **(**$\mathbf{WK^{-1}m^{-2}}$**)**	1.072	1.097	1.126	1.169	1.140	1.110
SWC **(%)**	21.580	33.890	34.492	29.551	33.085	29.417
β	1.501	1.589	2.039	1.098	1.191	1.383
NEE **(**μ**mol C** $\mathbf{m^{-2}s^{-1}}$**)**	-2.204	-2.843	-3.670	-6.120	-5.376	-4.148

* From 2009 to 2012, crop rotation at NE2 changed to continuous maize.

towers, which measure fluxes of energy, momentum, water, and carbon dioxide. The footprint of these towers can cover a longitudinal length of 100–2000 m depending on atmospheric conditions and sampling height (Baldocchi et al., 2001), which for this study is assumed to be 3 m at all sites. These specifications provide clear spatial boundaries in which sustainability can be evaluated for these ecosystems or agroecosystems. Long-term availability of observational data is a limitation in the broad application of this metric. Most EC tower sites only measure net radiation, which means that the incoming and outgoing shortwave and longwave radiation components have to be estimated using empirical formulations from the net radiation measurement in order to calculate the associated entropy production. Some EC tower sites, such as those selected for this study, measure all radiation terms individually and these data are preferable for calculating entropy production.

Level 2 standardized files with gaps were obtained at the AmeriFlux website (`http://ameriflux.ornl.gov/`) for the above described sites and available years from 2002-2012. Data with gaps were selected in order to eliminate potential errors associated with gap-filling. The reader is referred to the Summary Reports where information is available on percent available data for each variable by year (for example, availability of ARM variables can be found at `ftp://cdiac.ornl.gov/pub/ameriflux/data/Level2/Sites_ByName/ARM_SGP_Main/with_gaps/ARM_SGP_Main_SummaryReport.htm`). Outliers for all variables at all sites were removed above the 99th percentile and below the 1st percentile. For sites ARM, KFS and KON, half-hourly data were converted to hourly to match the hourly timescale of the NE1, NE2 and NE3 site data.

2.2.2 Thermodynamic entropy production

Following methodology from Brunsell et al. (2011), thermodynamic entropy production (σ) was calculated for each site across all available site years. Given Earth's energy budget, net radiation at the land surface (R_n, Wm^{-2}) is the balance between incoming and outgoing radiation that is partitioned into turbulent transport of sensible heat (H, Wm^{-2}) and latent heat (LE, Wm^{-2}) fluxes and heat conduction into the ground by the soil heat flux (G, Wm^{-2}):

$$R_n = Q_S + Q_{L,in} - Q_{L,out} = H + LE + G + \varepsilon \tag{2.1}$$

where the absorbed solar radiation is represented by Q_S (Wm^{-2}) and the longwave radiation terms are represented by $Q_{L,in}$ (Wm^{-2}) for incoming and $Q_{L,out}$ (Wm^{-2}) for outgoing. A residual of the energy budget (ε) accounts for the energy terms that are not considered in the other fluxes, like photosynthesis, and the instantaneous rate of heating. The inclusion of this term is important because energy balance closure rarely occurs with EC measurements (Wilson et al., 2002).

Thermodynamic entropy production can be calculated using R_n or the individual radiation terms, when available. For this study, we computed σ from both R_n for all sites and from the individual radiation terms for all sites except KON, where there is only a net radiometer. To obtain σ using R_n, the incoming and outgoing shortwave and longwave radiation components are derived from the R_n measurement as outlined below. This requires invoking the Monin-Obukhov similarity theory from air temperature, heat flux, and local stability (Foken, 2006) to derive the surface temperature (T_{sfc}, K), which is the temperature at the land surface disregarding any directional aspect of solar radiation (Wu, 2010) and assumed to be identical to the aerodynamic temperature (T_o, K):

$$T_o = T_a + \frac{H}{\kappa \rho c_p u_*}\left(\ln\left(\frac{z-d}{z_m}\right) + \psi_H\right) \tag{2.2}$$

where T_a is the air temperature at the measurement height (z) assumed to be 3 m at all sites, κ is the von Karman constant (0.4), ρ is the air density, c_p is the specific heat capacity, u_* is the friction velocity, d is the displacement height (2/3 height of canopy), and z_m is the aerodynamic roughness length ($(1/10)d$). A stability correction ψ_H is applied for unstable atmospheric conditions:

$$\psi_H = -2\ln\left(\frac{1+(1-16(z/L))^{1/2}}{2}\right) \tag{2.3}$$

where L is the Obukhov length.

The longwave radiation from the atmosphere ($Q_{L,in}$, Wm^{-2}) is calculated from the measured air temperature (T_a, K) using the empirical formulation of Brutsaert (1975):

$$Q_{L,in} = 0.552 e_a^{(1/7)} 5.67e^{-8} T_a^4 \tag{2.4}$$

where e_a is the actual vapor pressure in millibars.

The outgoing longwave radiation ($Q_{L,out}$, Wm^{-2}) is computed using the Stefan-Boltzmann Law with a constant surface emissivity assumed to be 0.95, and T_o assumed to be equal to T_{sfc}:

$$Q_{L,out} = 5.67e^{-8} 0.95 T_o^4 \tag{2.5}$$

The absorbed solar radiation (Q_S, Wm^{-2}) can then be computed as the difference between the measured R_n and the incoming and outgoing longwave radiation:

$$Q_S = R_n + Q_{L,out} - Q_{L,in} \tag{2.6}$$

with a minimum value of zero. Although the use of the aerodynamic temperature as the surface temperature introduces some complications, the two temperatures are related (Stewart et al., 1994). Any errors in the calculation of the radiation terms will propagate to the calculations of σ.

Values for Q_S and $Q_{L,in}$ that were either derived from R_n as outlined above or measured by a four-way radiometer, were then employed to calculate σ. Entropy production associated with solar radiation (σ_{Q_S}, WK^{-1}m^{-2}) is computed from the fluxes per unit area by:

$$\sigma_{Q_S} = Q_S \left(\frac{1}{T_{sfc}} - \frac{1}{T_{sun}} \right) \tag{2.7}$$

where the assumed temperature of 5780 K for the sun (T_{sun}) is constant and T_{sfc} is the same as T_o. The entropy production associated with the absorption of longwave radiation and degradation to heat (σ_{Q_L}, WK^{-1}m^{-2}) is:

$$\sigma_{Q_L} = Q_{L,in} \left(\frac{1}{T_{sfc}} - \frac{1}{T_{atm}} \right) \tag{2.8}$$

where the temperature of the atmosphere (T_{atm}, K) is computed by inverting the Stefan-Boltzmann law. To arrive at total entropy production (σ, WK^{-1}m^{-2}), the entropy terms for solar and longwave radiation are summed:

$$\sigma = \sigma_{Q_S} + \sigma_{Q_L} \tag{2.9}$$

Entropy production related to mass fluxes of water and carbon are not directly considered in these calculations. In addition, this study focuses only on thermal entropy production and omits the contribution of radiation pressure, which constitutes one third of the radiative entropy production factor of 4/3 (Goody & Abdou, 1996). A complete review of the entropy budget can be found in (Peixoto et al., 1991).

For all available Level 2 data (with gaps) at each AmeriFlux site between 2002 and 2012, hourly values of measured variables (u_*, T_a, H, LE, G, R_n, Q_S, $Q_{S,out}$, $Q_{L,in}$, $Q_{L,out}$, relative humidity (RH, %), pressure (P, kPa), vapor pressure deficit (VPD, kPa), SWC, and NEE) and calculated variables (ρ, L, ψ_H, T_{atm}, and T_o or T_{sfc}) were averaged across all days in a month to provide a daily cycle for each month. Only seven monthly diurnal cycles were missing across all 54 site years due to missing data and were, therefore, omitted from the analysis.

Thermodynamic entropy production was calculated from the average monthly diurnal cycle from R_n and from individual solar and longwave radiation terms at each site except KON, and the respective values of σ were compared (Figure 2.2). Since correlations were high, the analysis employed values of σ calculated from R_n in order to include the KON site years. Values from daylight hours of 9:00 a.m. to 4:00 p.m. were selected to analyze σ across sites and within land cover type with variables of SWC, β as an indicator of energy partitioning, and NEE of carbon. These monthly daytime averages, initially for all available years of data at each site and then for 2008-2012, define the temporal boundaries of this sustainability assessment.

Regression tree analysis was used to provide increased understanding of processes and thresholds contributing to entropy production. Using the rpart library in R (Therneau et al., 2010), regression models were built recursively and represented as binary trees. The regression trees were pruned with the 1 - standard error (SE) rule of Breiman et al. (1984), where the largest value of the complexity parameter (cp) with a cross-validation relative error (xerror) within one standard deviation of the minimum is selected. This value can be identified using the plotcp function and selecting the first cp value that lies below the 1 - SE dashed line. Mean standard error (MSE) values at each node were computed and used to evaluate the error in trees associated with entropy production for rainfed and irrigated cropland sites and grassland sites.

2.3 Results

Results indicate varying levels of thermodynamic entropy production and sustainability at each site related to interannual, seasonal, or extreme changes in water and carbon cycling given specific land cover and land management practices. The mean value of all monthly daytime observations for all site years is shown in Figure 2.1. These values indicate that σ varies from lowest to highest at ARM, KFS, NE3, KON, NE2, and NE1. Although this could be related to a latitudinal gradient of incoming solar radiation, it is opposite of what is expected (Peixoto et al., 1991). This is a first indication that σ is not soley determined by incoming solar radiation, but also by how the land surface or vegetation dissipates that energy based on other natural and anthropogenic inputs.

Across the entire time series, the most negative mean value of NEE indicating the largest

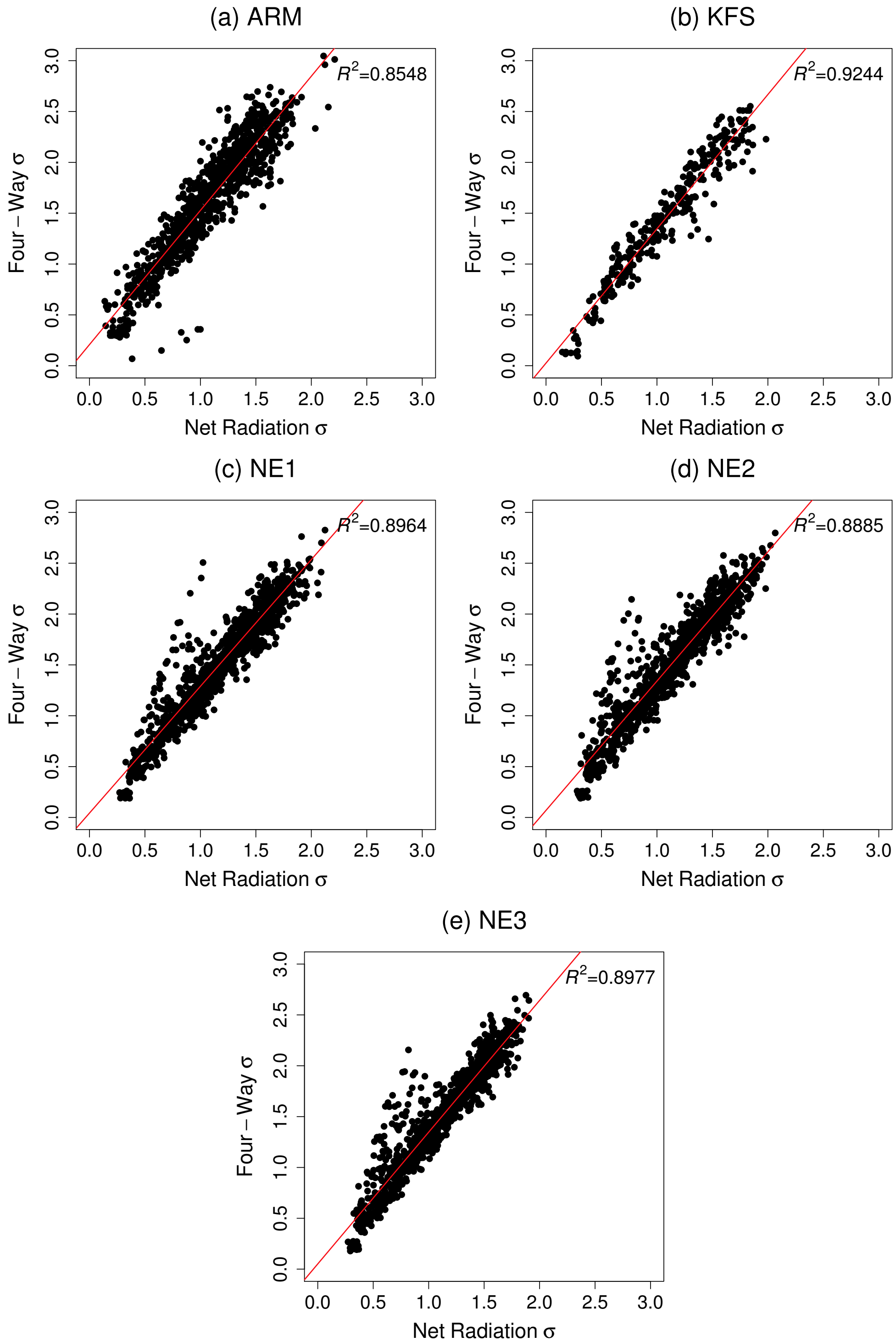

Figure 2.2: Correlations between σ from R_n and σ from individual radiation terms obtained by a four-way radiometer for (**a**) ARM, (**b**) KFS, (**c**) NE1, (**d**) NE2, and (**e**) NE3 using average monthly daytime (9am-4pm) values for all years available at each site.

overall carbon sink occurs at the NE1 site where there is the greatest mean value of σ (Table 2.1). Likewise, the largest mean value of NEE or the site which is the smallest carbon sink or greatest carbon source is the ARM site where the mean value of σ is at its lowest. Other values in Table 2.1 indicate that the rainfed cropland sites of ARM and NE3 have the lowest mean values of SWC, followed by the irrigated cropland sites of NE1 and NE2, and then the grassland sites of KFS and KON with the highest SWC. β values give us an indication of energy partitioning between H and LE at each site. All mean β values are above 1, which signifies that on average H is higher than LE for all sites throughout the daytime hours of the years examined. However, cropland sites have lower β values than grassland sites, indicating the greater role of LE throughout the year and especially during the growing season.

To further explore the differences between sites on a monthly scale, we examined time series of diurnal cycles of daytime hours for years 2008-2012 (Figure 2.3). The dashed red lines identify key values for each variable which help distinguish between diurnal, monthly, seasonal and annual patterns amongst sites. For example, σ values above 1.5 $\mathrm{WK^{-1}m^{-2}}$ (75th percentile) occur primarily in the summer months at all sites. SWC at irrigated cropland and grassland sites rarely falls below a value of 25% (25th percentile) with the exception of years 2011 and 2012 for rainfed cropland and grassland sites. As outlined above, whenever β is below 1, LE is greater than H which means that the energy partitioning at the surface is favoring dissipation of heat through LE. The opposite is true when β values are above 1. Similarly, values of NEE above 0 indicate a carbon source, while values below 0 indicate a carbon sink.

Compared to irrigated cropland sites, rainfed cropland sites ARM and NE3 have lower σ, higher β values, and larger NEE values making them smaller carbon sinks. The ARM site is the most arid with the lowest SWC and β values that rise throughout most months of the 2012 drought. At NE3, the NEE values vary according to the maize-soybean rotation from year to year. Across rainfed cropland and grassland sites, the drought of 2012 is seen in lower SWC values. The irrigated cropland sites maintain SWC values around 25% in 2012, but the σ values for NE1 and NE2 are especially elevated during this time.

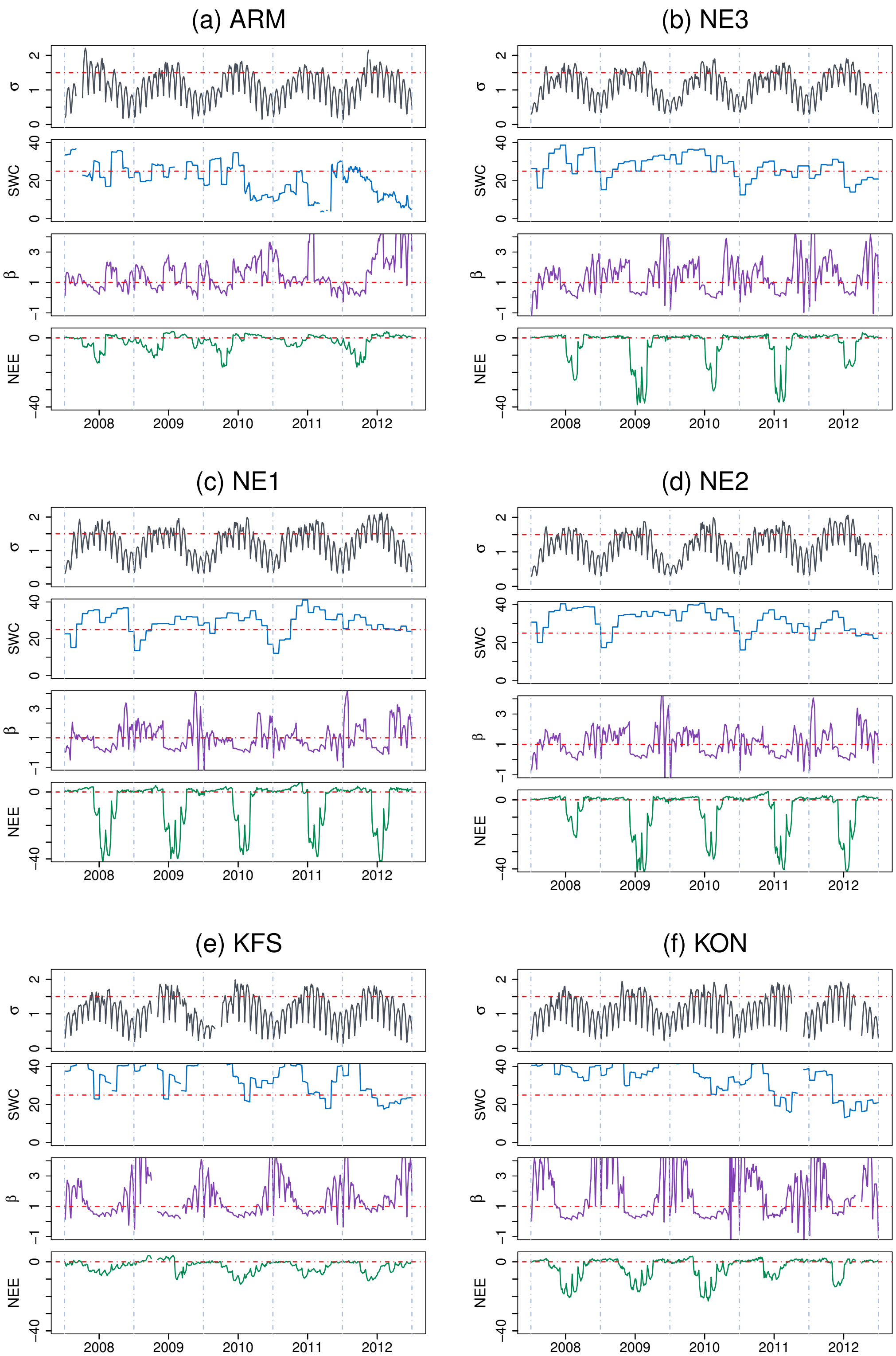

Figure 2.3: Monthly average daytime (9am-4pm) values of σ (WK^{-1}m^{-2}), SWC (%), β, and NEE (μmol C m^{-2}s^{-1}) for 2008-2012 at rainfed cropland sites (**a**) ARM and (**b**) NE3, irrigated cropland sites (**c**) NE1 and (**d**) NE2, and grassland sites (**e**) KFS and (**f**) KON.

Table 2.2: Based on time series shown in Figure 3, this table exhibits the mean and max annual values of σ ($WK^{-1}m^{-2}$), the number of times throughout a year that σ is above 1.5 $WK^{-1}m^{-2}$, and the mean growing season (GS) values of σ for April through September. Mean GS values for the ARM site are not shown due to the difference in growing season.

		ARM	**KFS**	**KON**	**NE1**	**NE2**	**NE3**
2008							
	Mean Ann σ	1.132	1.069	1.077	1.195	1.102	1.104
	Max Ann σ	2.211	1.712	1.798	1.936	1.726	1.772
	$\sigma > 1.5$	21	9	12	29	18	18
	Mean GS σ		1.270	1.276	1.442	1.349	1.354
2009							
	Mean Ann σ	1.072	1.119	1.125	1.184	1.135	1.130
	Max Ann σ	1.799	1.859	1.936	1.958	1.877	1.813
	$\sigma > 1.5$	15	21	21	29	24	23
	Mean GS σ		1.449	1.363	1.455	1.421	1.395
2010							
	Mean Ann σ	1.092	1.102	1.187	1.195	1.102	1.086
	Max Ann σ	1.833	1.983	1.915	1.978	1.987	1.901
	$\sigma > 1.5$	21	22	26	28	18	13
	Mean GS σ		1.394	1.367	1.465	1.401	1.375
2011							
	Mean Ann σ	1.078	1.088	1.169	1.232	1.204	1.120
	Max Ann σ	1.730	1.862	1.897	1.912	1.897	1.782
	$\sigma > 1.5$	15	18	18	29	30	17
	Mean GS σ		1.340	1.340	1.502	1.471	1.378
2012							
	Mean Ann σ	1.163	1.111	1.131	1.312	1.277	1.177
	Max Ann σ	2.153	1.811	1.923	2.123	2.066	1.904
	$\sigma > 1.5$	21	21	20	37	30	27
	Mean GS σ		1.361	1.423	1.651	1.580	1.469

To determine the role of the 2012 drought on σ at each site, Table 2.2 outlines the variation in mean and maximum annual values, mean growing season values, and the number of times σ is elevated above 1.5 $WK^{-1}m^{-2}$ as shown in Figure 2.3. While the Nebraska sites experience an increase in mean and maximum annual σ, mean growing season σ, and the number of times σ is above 1.5 $WK^{-1}m^{-2}$ for 2012, the ARM and grassland sites exhibit little variation from previous years. Of the Nebraska sites, the rainfed NE3 site still has lower σ values than irrigated sites NE1 and NE2 in 2012.

Figure 2.4 is a pairwise, Pearson correlation matrix of the time series values plotted in Figure 2.3. The values of σ across sites are most correlated with the values of NEE, with 2008-2012 correlations between -0.46 and -0.51 at all sites except ARM where the correlation is -0.11 (Figure 2.4). Examining cross-site correlations helps to distinguish between sites that may have similarities in land management practices or energy inputs that can contribute to σ. Values of σ at all sites are well correlated with each other, and especially strong correlations are seen amongst the Nebraska cropland sites and between the Kansas grassland sites further supporting the role of incoming solar radiation and vegetation fraction (Brunsell et al., 2011). The Nebraska sites show strong positive correlations between themselves for SWC, β, and NEE values as well. Other strong positive correlations worth mentioning include the SWC at irrigated NE2 and rainfed NE3 (R^2=0.9), the SWC at KFS and KON (R^2=0.81), and the β values between KFS and KON (R^2=0.81). A pattern related to the ARM site emerges from the correlation matrix; mainly, σ and NEE are negatively correlated at all tower sites, but ARM exhibits much smaller negative correlations, and there are positive correlations between β and NEE at all sites except ARM.

The regression trees in Figure 2.5 attempt to clarify the role of SWC, β, and NEE in thermodynamic entropy production and identify important thresholds for these variables. Mean squared error (MSE) values for all nodes in all three trees are shown in Table 2.3, indicating that the root node for all trees have similarly low error allowing them to be compared.

In rainfed cropland sites (Figure 2.5a), higher σ values of 1.2 WK^{-1}m^{-2} and greater (50% of data) occur when a site is a net source above 0.9 μmol C m^{-2} or a net sink below -4.5 μmol C m^{-2}. Values of σ around 1 WK^{-1}m^{-2} occur with NEE between -1.5 and 0.9 μmol C m^{-2}s^{-1} and β values greater than 1.4, while lower σ values around 0.81 WK^{-1}m^{-2} occur when β values are less than 1.4. Surprisingly, SWC is not shown to be a primary variable in distinguishing between levels of σ for rainfed cropland sites.

For irrigated cropland sites (Figure 2.5b), the highest σ values of 1.7 WK^{-1}m^{-2} occur when NEE is below -34 μmol C m^{-2}s^{-1} corresponding to a large uptake of carbon. As long as NEE indicates a net sink between -34 and -7.9 μmol C m^{-2}s^{-1}, σ remains high around 1.4 WK^{-1}m^{-2}. Lower σ values between 0.81 and 1.2 WK^{-1}m^{-2} (62 % of data) occur as NEE transitions from a net sink of -7.9 μmol C m^{-2}s^{-1} to a net source of 1.3 μmol C m^{-2}s^{-1} and above. Similar to rainfed cropland sites, higher values of σ around 1.1 WK^{-1}m^{-2} occur when β is larger than 1.4, while lower σ values around 0.81 WK^{-1}m^{-2} tend to happen when β is less than 1.4. SWC is also

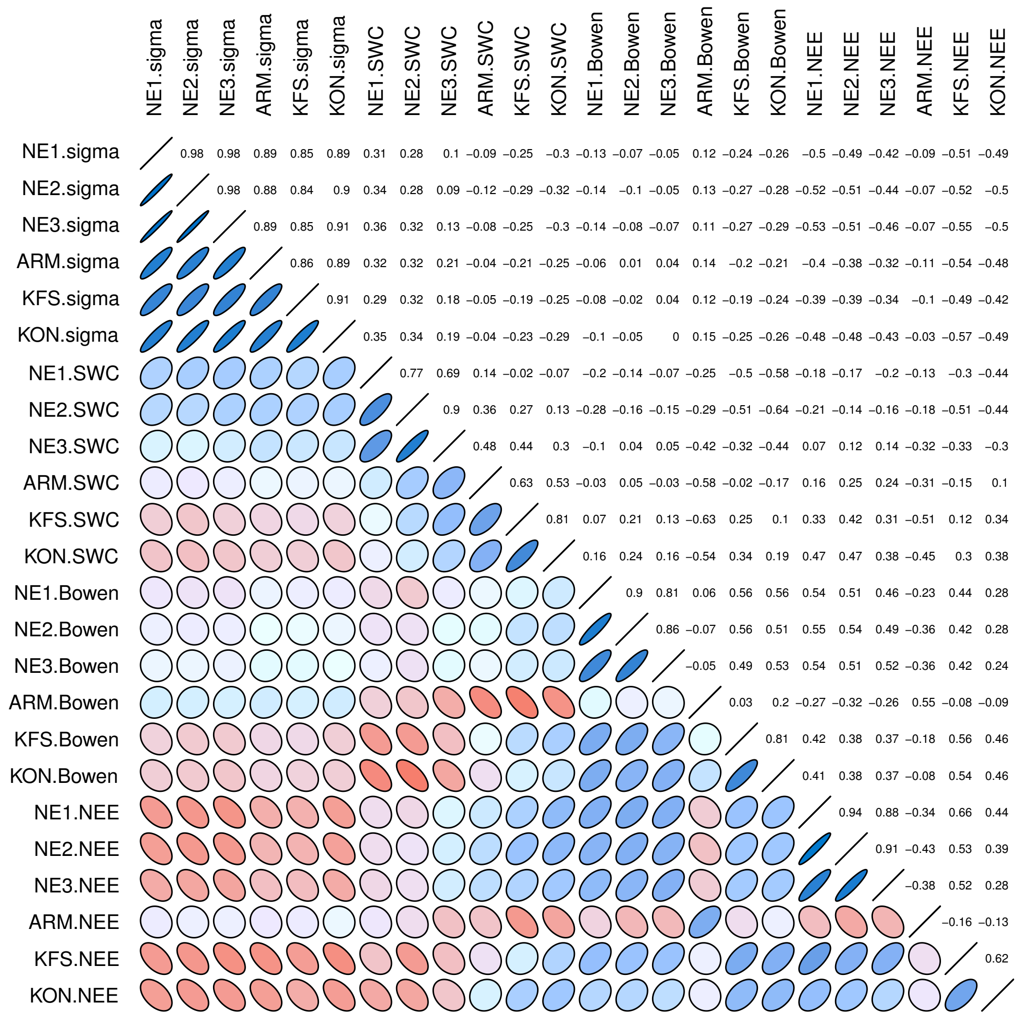

Figure 2.4: Correlation matrix of σ, SWC, β, and NEE values in Figure 5.

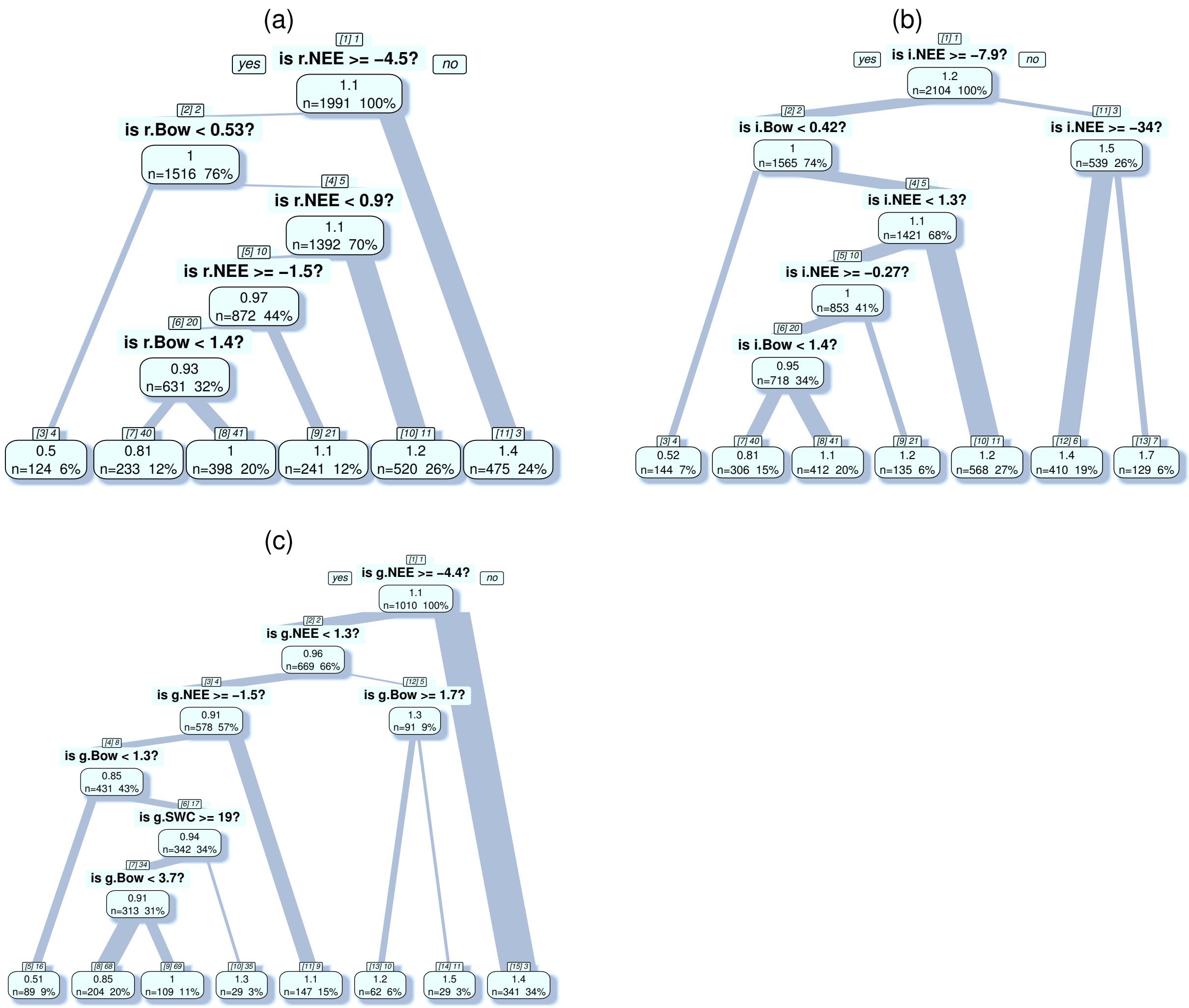

Figure 2.5: Regression trees for σ from SWC, β, and NEE values at (**a**) rainfed cropland sites ARM and NE3, (**b**) irrigated cropland sites NE1 and NE2, (**c**) and grassland sites KFS and KON.

Table 2.3: Mean standard error (MSE) values at each node (number in brackets) for Figure 5a-c.

Node	[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]	[9]	[10]	[11]	[12]	[13]	[14]	[15]
(a)	0.154	0.148	0.049	0.131	0.135	0.135	0.143	0.116	0.117	0.093	0.085				
(b)	0.171	0.146	0.053	0.126	0.124	0.111	0.102	0.091	0.121	0.101	0.075	0.073	0.026		
(c)	0.164	0.134	0.122	0.114	0.068	0.088	0.076	0.092	0.031	0.099	0.099	0.084	0.032	0.110	0.094

not a primary variable in distinguishing between levels of σ for irrigated cropland sites.

When σ of grasslands is examined as a function of SWC, β, and NEE, σ values on the order of 1.4 WK^{-1}m^{-2} (34% of data) can be explained by NEE that is less than -4.4 μmol C m^{-2}s^{-1} (Figure 2.5c). Lower σ values of 1.1 WK^{-1}m^{-2} and less are explained when NEE is between -4.4 and 1.3 μmol C m^{-2}s^{-1}. In addition, some of the lowest σ values of 0.85 WK^{-1}m^{-2} occur when NEE is between -1.5 and 1.3 μmol C m^{-2}s^{-1}, SWC is equal to or greater than 19%, and β is less than 3.7. If SWC dips below 19%, greater σ of 1.3 WK^{-1}m^{-2} may occur.

2.4 Discussion

Selecting meaningful indicators for agroecosystem sustainability begins with trials regarding the usefulness of metrics to identify the most pertinent stressors and underlying processes within certain spatial and temporal boundaries. While agroecosystem sustainability in light of climate science calls for optimization of location-specific practices, geographic variability of biophysical sustainability remains poorly understood.

In this case study, variability of σ across AmeriFlux cropland and grassland sites indicates varying levels of sustainability given land cover, land management, and the climatic impact of drought. While detailed information on crop type, tillage, fertilization, pest control, irrigation, and harvest may allow for a robust comparison between sites and within site years, this information may not be necessary given AmeriFlux data for SWC, σ, and NEE.

Agriculture in the US uses energy directly through fuel and electricity for the operation of machinery and irrigation on the farm, and indirectly through the utilization of fertilizers and chemicals produced off the farm (Schnepf, 2004). Alternative agricultural management practices such as continuous no-till can reduce both direct and indirect energy use and offer additional benefits to agroecosystems, including increased water infiltration and increased carbon sequestration for climate change mitigation (Derpsch et al., 2010; Kell, 2011; Lal, 2011). If the key to achieving greater agroecosystem sustainability is to minimize σ given energy inputs (Steinborn & Svirezhev, 2000; Svirezhev, 2000; Patzek, 2008), such as those related to irrigation and fertilizer, our results can help identify when this occurs.

Average σ is the highest at irrigated cropland sites NE1 and NE2 that both grow maize requiring large inputs of fertilizer. This result supports the findings of Patzek (2008) that maize monocultures are not sustainable regardless of tillage practice. Compared to other cropland sites, NE1 and NE2 also have an average SWC that is higher, an average σ that is lower, and an average NEE that is the most negative corresponding to the greatest sinks of carbon (Table 1). However, NE2 has average values that are slightly lower for σ, higher for SWC and σ, and less negative for NEE. These differences between the NE1 and NE2 sites are most likely related to the maize-soybean rotation occurring until 2009 at NE2, where years in soybean production are associated with lower fertilizer applications and lower vegetation fraction (Verma et al., 2005). While we are only examining daytime values for NEE, our finding that NE1 is a greater carbon sink than NE2 is corroborated by Verma et al. (2005). From 2001-2004, NE1 was found to be carbon neutral to a slight carbon source, and NE2 was a moderate carbon source, primarily due to increased rates of respiration with higher SWC during years when soybeans were planted (Verma et al., 2005).

Rainfed cropland site NE3, which also has a maize-soybean crop rotation, exhibits the lowest average σ, lowest average SWC, highest average β, and least negative average NEE of the Nebraska sites (Table 1). Considering the lower inputs of fertilizer associated with maize-soybean rotation similar to NE2 (Verma et al., 2005) and the lack of irrigation, it is not surprising to see that it has the lowest σ and could be considered more sustainable than both NE1 and NE2. SWC at NE3 is actually well correlated with the irrigated NE2 site (Figure 4) indicating the importance of crop type on soil moisture. However, the lower σ accompanies a NEE that is indicative of the smallest sink of carbon of all Nebraska sites across daytime hours of all years. From 2001-2004, Verma et al. (2005) found the site to be carbon neutral, and grain yield was smaller mostly because of differences in plant densities which are lower on rainfed plots (Verma et al., 2005). Without the data to quantify the crop yield and biomass exported off-site during 2002-2012 across all sites, the relationship between σ and NEE in our study remains relative to on-site, daytime production. Excluding the nighttime values of NEE from our analysis disregards the role of respiration.

The ARM rainfed cropland site in Oklahoma exhibits the lowest average σ, the lowest average SWC, the highest average σ, and the least negative average NEE of all cropland sites (Table 1). The primary crop grown at this site is winter wheat, which requires less fertilizer inputs than maize but more than soybeans (USDA, 2013) and has a growing season from approximately October to May. Similar to site NE3, the ARM site is rainfed so there are no anthropogenic inputs of energy related to irrigation. The influence of the lack of water on SWC, σ, and NEE is especially pronounced during the Southern Plains drought in 2011 when Oklahoma experienced extremely dry conditions (Karl et al., 2012). During this time, the ARM site does not exhibit a rise in mean or maximum annual σ, though it does exhibit a slight elevation in 2012 (Table 2). This is likely due to the timing of both the 2011 and 2012 drought in relation to the growing season for winter wheat.

Kansas grassland sites KFS and KON have an average σ higher than the ARM site, which may again be due to winter wheat production and the vegetative fraction during certain times of the year Brunsell et al. (2011). The Kansas sites also exhibit the greatest average SWC and β, and NEE values that are higher than the ARM site but lower than the Nebraska sites. KFS and KON are well correlated for SWC and σ (Figure 4), but the KON site has unexpectedly higher σ. Increased σ due to the annual burns in April and associated green-up may indicate that annual burns at KON are less sustainable than five-year burns at KFS. Maintaining a native prairie by burning every year at KON might actually be putting more stress on the system by forcing it to remain as a tallgrass system and not allowing it to evolve toward a site with woody encroachment, which is what is happening on neighboring Konza Prairie plots with different burn regimes Cochran et al. (2013). In addition, missing data in months with lower σ may be skewing the KON average to be higher than both KFS and the rainfed cropland site of NE3.

According to Figure 5c, when σ values are low in grasslands a set of conditions occur where

NEE is between -1.5 and 1.3 μmol C $m^{-2}s^{-1}$, SWC is greater than or equal to 19%, and β is less than 3.7. In other words, at times when grasslands are basically carbon neutral, SWC can be low, and H can be high or low, Kansas grasslands exhibit low σ values indicating greater sustainability. This time of greater sustainability may correspond to periods when the grassland is not experiencing rapid changes in water or carbon cycling and is closer to thermodynamic equilibrium. So, given the low values of σ for grasslands during the 2012 drought (Table 2), we can imagine that even if a grassland is water or heat stressed, as long as it remains carbon neutral, it will remain sustainable. Furthermore, the set of conditions seen in Figure 5c may be a seasonal signal of perennial grassland species, many of which are C4, which highlights the importance of evaluating growth phases for grassland and cropland species and their contribution to σ, similar to critical climate periods (Craine et al., 2012).

While σ was elevated at all cropland sites during 2012, an extreme drought year for the Central US (Karl et al., 2012), it was especially elevated during the growing season at irrigated sites NE1 and NE2 (Table 2). Above normal maximum and minimum temperatures in Spring and Summer of 2012 broke previous records from 1934 (Karl et al., 2012). July 2012 was the hottest month in the instrumental record and by August 2012, the US Drought Monitor categorized the drought in Kansas, Nebraska, and Oklahoma as extreme to exceptional. During this time, NE1 and NE2 exhibited higher diurnal σ values compared to previous years and compared to all other sites in 2012. One explanation for the elevated σ is that extra inputs of irrigation needed to maintain maize production during the 2012 drought resulted in increased gradients of temperatures and fluxes of heat influencing σ. If we relate this to the negative feedback loop in the evolutionary framework described in the Introduction, we can see this as an example of increased σ associated with work done in "deviation-countering" processes for maintaining similar levels of NEE to non-drought years. On the other hand, σ at the rainfed NE3 site during the same stressful disturbance was not elevated, suggesting that this agroecosystem is not continually working to overcome gradients but has instead entered the positive feedback loop where NEE is diminishing, the selection of traits is underway, and adaptation will follow.

Above normal temperatures are often associated with periods of water stress, which can also lead to reduced evapotranspiration and LE corresponding to increased β. Regression trees for cropland sites shown in Figures 5a and 5b identify a potential threshold for σ of 1.4. Above this threshold where energy partitioning favors H, σ values remain higher around 1.0 $WK^{-1}m^{-2}$ and greater. Below the threshold where energy partitioning favors LE, σ values remain low around 0.8 $WK^{-1}m^{-2}$ and lower. The lowest values of σ around 0.5 $WK^{-1}m^{-2}$ occur when β is below 0.53 for rainfed croplands and below 0.42 for irrigated croplands. Considering that greater entropy production is usually associated with LE rather than H as entropy is transferred out of a system (Peixoto et al., 1991), the reader is reminded that this study focuses only on production associated with the absorption of solar and longwave radiation.

Therefore, our findings suggest that at times when σ is low in cropland sites, increased LE corresponds with a greater dissipation of heat, a reduction of temperature gradients, and a greater transfer of entropy from the system. Conversely, at times when σ is higher, a greater H corresponds with temperature gradients between the surface and the atmosphere that are steeper resulting in higher σ and entropy that is being transferred at a much slower rate (Peixoto et al., 1991). Thus, from a biophysical perspective, these results provide further support that an agricultural system that is closer to thermodynamic equilibrium is more sustainable.

Since the values of σ across sites are most correlated with the values of NEE (Figure 4), we infer that energy inputs that drive NEE and associated processes of photosynthesis, respiration, and evapotranspiration also influence sustainability by driving croplands away from thermodynamic equilibrium. Therefore, like grasslands, croplands may experience a tradeoff between carbon uptake and sustainability. Moreover, given the FAO definition of climate-smart agriculture (Lipper et al., 2010), it is unclear whether rainfed cropland sites ARM and NE3, which are most sustainable according to our findings, would fulfill the criteria for GHG mitigation. In drought conditions like 2012 when a rainfed cropland site might be less of a carbon sink, irrigation inputs to maintain a certain level of NEE in cropland sites like NE1 and NE2 may result in an overproduction of entropy as discussed by Svirezhev (2000). In this context, a more sustainable agroecosystem may be one that is not maximizing carbon uptake, like we might be inclined to do for climate change mitigation and food security purposes. If we are to maintain a carbon sink, even in drought conditions, the goal should be to do so without increasing entropy production. Quantifying the entropy production of drought-tolerant crops, such as sorghum, would be an interesting comparison to the crops included in this study and an obvious next step in assessing biophysical sustainability for agroecosystems. Overall, the difference between rainfed and irrigated croplands in Figures 5a and 5b indicates that mitigation of carbon in agroecosystems becomes less sustainable when we hit a σ threshold above 1.4 $\mathrm{WK^{-1}m^{-2}}$.

As we examine our results and consider improvements for continued analyses, we recognize that the availability of AmeriFlux data may contribute to the reliability of results. For example, differences in percent available data between sites and between site years will influence the mean monthly diurnal cycle and introduce error between months to years for any given site. Future studies utilizing additional AmeriFlux or FLUXNET site years could increase the reliability of σ as a sustainability metric. The availability of reanalysis data, such as the ECMWF ERA-Interim product, could also allow for a broader application of this biophysical indicator on a global scale. In addition, detailed land-use histories for AmeriFlux sites are needed to better evaluate influences of anthropogenic energy inputs, like fertilizer. We need to more thoroughly explore how σ relates to crop and biomass yield, while accounting for carbon exports related to harvest, fire, and mowing. Such information will also help to incorporate social and economic dimensions of sustainability, which may further differentiate between sites.

2.5 Conclusions

Based on our assessment of Central US croplands and grasslands, the entropy production (σ) metric can provide us with a good indication of whether a system is moving away from or toward thermodynamic equilibrium and sustainability. Except for the ARM site in Oklahoma, σ is most correlated with NEE, which highlights the role of vegetation on the equilibrium conditions of the land surface (Brunsell et al., 2011). Elevated σ occurs in irrigated and rainfed agroecosystems that are stressed by drought conditions in 2011 and 2012. Though irrigation may allow agroecosystems to maintain non-drought levels of NEE, as exemplified by site NE1 with irrigated maize, the associated energy inputs result in higher σ above a threshold of 1.4 $\mathrm{WK}^{-1}\mathrm{m}^{-2}$ and a move away from biophysical sustainability. In other words, more work is required to overcome gradients created by an irrigated system in drought conditions, and we suggest that the system then remains "forced" in a negative feedback loop where it is not experiencing changes that can result in adaptation at the ecosystem scale. This decrease in sustainability has implications for increased agricultural production and food security in areas that depend on irrigation and fertilizer use. Therefore, our results suggest that long-term GHG mitigation for climate-smart agriculture is more sustainable in rainfed agroecosystems.

This case study is a first attempt to unify theories of complex ecosystem dynamics with thermodynamic laws to better define sustainability within the physical geography community. Based on the mechanisms involved in Wallace's evolutionary theory at the ecosystem scale, we present methodology to quantify a move away from or toward adaptation, resilience, and sustainability. By further comparing σ with SWC, the Bowen ratio, and NEE of carbon, we have also identified processes underlying moves toward or away from sustainability, which can hopefully render the metric more useful.

Biogeographers, climatologists, and landscape ecologists alike, may be interested in further relating how the evolution of spatial interactions and processes involved in ecosystem development and agroecosystem production influence the geographic variability of sustainability. With available data from eddy-covariance towers, satellite remote sensing instruments, and reanalysis output, opportunities to empirically study these interactions and processes exist around the globe. Physical geographers possess the tools to advance theoretical concepts, methodologies, and analyses to continue addressing the "negative space" that characterizes the evolution of spatial interactions for sustainability.

Acknowledgements

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Chapter 3

Biophysical Metrics for Detecting More Sustainable Urban Forms at the Global Scale

Ferdouz V. Cochran and Nathaniel A. Brunsell

Abstract

To test metrics for rapid identification and global evaluation of more sustainable urban forms, we examine the configuration of the São Paulo Metropolitan Region (SPMR) in Brazil using satellite remote sensing data and landscape metrics. We adopt principles from landscape ecology and urban planning to evaluate urban heterogeneity and morphology that may constitute more sustainable urban forms, including connectivity, density, complexity (mixed land uses), diversity, and greening. Using 2-D wavelet multiresolution analysis and satellite-derived fractional vegetation cover (Fr), the variability of landscape metrics from Landsat (30 m) to MODIS (1 km) scales are investigated. According to our findings, metrics of Patch Density and Landscape Shape Index can be used at the 1-km scale to asses density and geometric complexity of urban form. With the addition of MODIS land surface temperature (LST) data, available at high temporal resolution, a move away from or toward more sustainable urban forms is defined. As the complexity and density of finer-scale urban characteristics are related to climatic impacts at the neighborhood scale, sustainability assessments may be more attainable across urban areas.

3.1 Introduction

An urban classification system at the global scale is needed for input in climate models (Jackson et al., 2010) and for monitoring of environmental sustainability. The ability to rapidly classify urban intensity, density, or morphology on a global scale using satellite remote sensing data has eluded researchers for over a decade, though great strides have been made in classifying individual cities using high resolution imagery (Weng, 2012; Schneider et al., 2010). Methods for classification of impervious surface areas (ISA), for example, include pixel-based, sub-pixel based, object-oriented algorithms and artificial neural networks (Weng, 2012). Most of these methods were developed for a resolution of 10-100 m, and the availability of LiDAR data, in particular, is shifting research toward finer scales. Yet, broad application of high resolution urban classifications remains limited by the availability of data (such as cloud-free Landsat scenes in certain regions), computing power, and cultural or regional biases that skew classifications. While more research may be needed on the spectral, geometric and temporal aspects of urban mapping at finer scales (Weng, 2012), continued exploration of urban forms at the coarser, 1-km scale and how they relate to the finer scales may aid in more generalized studies where temporal resolution is key.

The movement toward more sustainable cities requires both broad temporal and spatial analyses, and Grimmond et al. (2010, pg. 259) identify a high need in climate change assessments "at the scale of cities to ensure that the signal of climate change is distinguished from the noise of natural variability." In the context of global climate change, and from a non-equilibrium or chaos perspective, the idea of "fail-safe" urban forms when it comes to sustainability is antiquated (Ahern, 2011). Mori & Christodoulou (2012) identify the need for a City Sustainability Index (CSI) that considers the triple bottom line for strong sustainability, incorporates leakage effects, and assesses developing to developed cities in an equitable manner. Strong sustainability relies on the continued availability of natural capital, whereas weak sustainability allows human capital to replace natural capital (Dietz & Neumayer, 2007). Within city boundaries, one of the most obvious ways to identify natural capital is through fractional vegetation cover related to green infrastructure which provides access to pervious surface areas (Andersson, 2006; Lehmann et al., 2014) and mitigation of urban heat islands (UHI) through shading and evapotranspiration (Oke, 1988; Smith & Levermore, 2008; Stone & Rodgers, 2001).

How vegetation relates to land surface temperatures (LST) continues to be investigated in remote sensing studies (Carlson & Arthur, 2000; Gillies et al., 1997; Li et al., 2011; Weng et al., 2004; Yuan & Bauer, 2007). A triangle-shaped scattergram typically results from plotting LST versus normalized difference vegetation index (NDVI) or fractional vegetation cover (Fr), and cold and warm edges of the triangle correspond to the wettest and driest pixels, respectively. What has been termed the "triangle method" has been used for estimating soil surface wetness and evapotranspiration from satellite imagery, and Carlson & Arthur (2000) and Carlson (2007) have showed how the temporal trajectory of pixels within the triangle can be associated with land use changes and urbanization. In addition, Gallo et al. (1993) used NDVI to evaluate urban and rural differences in minimum temperatures and found that NDVI approximated temperature variances more accurately than data on urban populations in the US. They suggested the approach as a method for consistent global evaluation of the urban heat island effect. Our study combines the knowledge of the triangle method, the relationship between LST and NDVI or Fr, with landscape metrics to evaluate the temporal trajectory of urban form.

Urban form can be defined as structural elements that make up a city, including natural features and open space, and the general pattern of building intensity and height (Lynch, 1982). Characteristics of urban form and how they relate to sustainability have been debated and quantified for some time (Burton et al., 2013; Jabareen, 2006; Shirowzhan & Lim, 2013; Stefanov & Netzband, 2005; Williams et al., 2000; Zhang & Guindon, 2006). Satellite imagery is an obvious tool for assessing urban sustainability around the world (Netzband et al., 2007), and one of the first formal indices created on a global scale was the Eco-Value Night Light Environmental Sustainability Index (Sutton, 2003). In addition, the role of landscape ecology in urban planning and the use of landscape metrics for evaluating more sustainable urban forms has been explored for over a decade (Huang et al., 2007; Leitao & Ahern, 2002; Renetzeder et al., 2010; Wu, 2008, 2009; Yang et al., 2014).

Urban forms that may be considered more sustainable are characterized by "urban pattern that is compact, pedestrian oriented, less autodependent, and not disaggregated into single, functional-use zones" (Duany & Talen, 2007). Duany & Talen (2007) proposed a transect approach for urban planning based on ecological theory, where a geographic cross-section of a city might reveal a continuum of human habitats with diminishing intensities from urban to rural that can satisfy all human needs. In transect planning, the attempt to eliminate sprawl discourages "urbanizing of the rural" or "ruralizing of the urban." Transect zones are influenced by principles of traditional neighborhood development (TND) where a sustainable neighborhood pattern is one that fulfills human needs for connectivity and diversity. A sustainable neighborhood is designed for humans, not automobiles.

In biophilic urbanism, a city's inhabitants' physical and mental health, work productivity, and social capital are improved by putting "nature first in its design, planning, and management" (Beatley & Newman, 2013; Beatley, 2009). Biophilic cities can contribute to urban sustainability on many levels, and green infrastructure associated with rivers, floodplains, wetlands and forests usually increase adaptive capacity when it comes to climatic impacts. As urban planners become more aware of the importance of urban ecosystem services, ways of quantifying these services by defining urban vegetation structure types and their associated micro-climatic effects are being investigated (Lehmann et al., 2014). Urban forms that exhibit connectivity and landscape heterogeneity are said to be essential for the provision of ecosystem services and long-term sustainability (Andersson, 2006).

Various definitions of sustainable urban form exist (Burton et al., 2013), and terms of compactness, complexity, connectivity, density, diversity, and greening have been repeatedly considered in their quantification (Burton et al., 2013; Jabareen, 2006; Shirowzhan & Lim, 2013; Williams et al., 2000). If we combine concepts from Duany & Talen (2007), Beatley & Newman (2013) and Jabareen (2006), over time there should be an increase in connectivity, diversity, and greening for improved urban sustainability. However, when it comes to density and compactness, Burton et al. (2013) conclude that we should look instead for various urban forms that are appropriate depending on scale and location. This recommendation is supported by the concern that policies calling for increased urban density for mitigation of greenhouse gases from vehicle miles traveled may be in conflict with other climate change adaptation strategies, such as floodwater management that may require greater areas of pervious surfaces depending on topography (Hamin & Gurran, 2009).

Therefore, the definition we propose for more sustainable urban forms is: Biophysical forms that maintain connectivity, complexity (diversity or mixed land use), greening, and scale- or location-specific levels of density (compactness or aggregation) that mitigate the UHI and allow for adaptation to climate change impacts. Given this definition, this study asks the question: How can MODIS satellite products be employed for long-term monitoring of biophysical metrics that relate to more sustainable urban forms at the 1-km scale? To answer this question, we undertake a case study of what a 1-km scale classification might look like and how it might relate to both finer scale heterogeneity and landscape metrics for monitoring more sustainable urban forms.

We employ an urban classification based on Landsat fractional vegetation cover (Fr) (Carlson & Ripley, 1997). Simplistically, Fr represents both a measure of surface structure and surface cover (Stewart & Oke, 2012). From a climate perspective, it is important to incorporate both of these categories to fully address the variations in airflow, albedo, surface radiation balances, and heat and moisture transport within urban areas that contribute to the UHI (Stewart & Oke, 2012). Our simple classification is used as a surrogate for the forthcoming local climate zone (LCZ) classification, one of the most promising urban classifications at the 30-m scale. The World Urban Database and Access Portal Tools (WUDAPT) project is making great progress toward this classification based on methodology using Landsat imagery (Bechtel & Daneke, 2012; Bechtel et al., 2015). Designed for the purpose of standardizing urban heat island (UHI) studies around the globe, LCZs define urban zones primarily based on their thermal properties. Stewart et al. (2013) have confirmed that building height and spacing, tree density, pervious surface fraction and soil wetness are major drivers of urban thermal differences.

The purpose of our study is to uncover a relatively simple and rapid way, given any urban classification system, to evaluate urban sustainability at the global scale. Scale, including the choice of system boundaries and methods for spatial aggregation, is a major hurdle in sustainability assessments (Burton et al., 2013; Mayer, 2008). In both urban and rural areas, information ascertained from various scales is crucial for the creation and implementation of policies for greater environmental sustainability (Renetzeder et al., 2010).

3.2 Study Region

This case study focuses on the São Paulo Metropolitan Region (SPMR) in southern Brazil, an agglomeration of 39 municipalities which includes at its center the city of São Paulo. São Paulo is Brazil's largest city, and the SPMR is home to 19.7 million people or approximately 10% of the national population (Haddad & Teixeira, 2015). It ranks fifth in the world as far as urban agglomerations (UN, 2014).

The Brazilian economy depends greatly on this urban area which contributes 19% of Brazilian GDP (Haddad & Teixeira, 2015). With climatic changes and an increase in the impervious surface area of the Upper Tietê River Basin, situated within the SPMR boundary, flood frequency has become greater in the last decade impacting regional, domestic and international trade markets (Haddad & Teixeira, 2015).

The carbon footprint of the SPMR, including direct emissions, was estimated to be 1.15 metric tons per person, smaller than the national average of 2.44 metric tons and the global average of approximately 4 metric tons (Sovacool & Brown, 2010). About 51% of emissions come from transportation, with automobile use being a primary contributor. Because most of the electricity used in the city is produced by hydroelectric plants, building and industrial energy use accounts for only 24% of emissions. Landfill waste accounts for 23% while agriculture and forestry contribute the remaining 2% of emissions.

In the last two decades, city managers and planners have focused on reducing GHG emissions from transportation and solid waste sectors. Restrictions on traffic have included exclusion of

20% of private automobiles during peak hours on weekdays. In addition, hybrid electric buses were added to public transportation fleets and private vehicles were converted to natural gas. An 11% reduction in GHG emissions was reported where methane release from solid waste in the Bandeirantes landfill has been used for power generation (de Oliveira, 2009).

In 2003, São Paulo joined the Cities for Climate Protection (CCP) program, part of the International Council for Local Environmental Initiatives (ICLEI) a transnational municipal network focusing primarily on climate change mitigation efforts. The city also joined the Energy Efficiency Program for the State of São Paulo and in June 2009 passed Municipal Law 14.933, which established a target of 30% reduction in GHG emissions by 2012 (D'Almeida Martins & da Costa Ferreira, 2011). The law also outlined construction and land use measures, though details on implementation and compliance may still be under development. Construction measures included energy efficiency, material quality, and greenspace guidelines for all new constructions. Land use measures called for increasing urban density in commercial and job centers to reduce commuting time and increasing greenspaces and trees throughout the city. A Master Plan for São Paulo created in 2002 reduced floor area ratio (FAR) in new constructions in parts of the city, and with a housing boom in 2005, a transition toward the payment of fees associated with building projects that exceeded new FAR regulations was observed (Sandroni, 2011).

Although certain mitigation measures may be in place, city planning is lacking when it comes to climate change adaptation measures, especially related to flooding and water scarcity (Ruijs et al., 2008). Environmental and socio-economic problems associated with rapid industrialization and population growth throughout the 20th century abound (Cohen, 2004). Despite recent social housing initiatives, where slums throughout the city are urbanized, over a quarter of the urban population lives in substandard housing with little to no access to clean water and sanitation services (Sandroni, 2011).

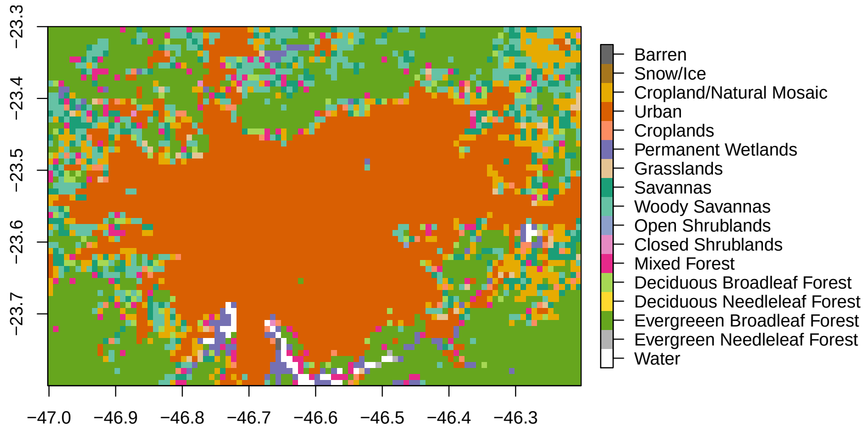

Figure 3.1: The urban extent of São Paulo Metropolitan Region (SPMR), Brazil from the the IGBP global vegetation classification scheme in the Moderate Resolution Imaging Spectroradiometer (MODIS) Land Cover Type product (MCD12Q1) aggregated to 1 km.

3.3 Methods

To extract the urban area of the SPMR for this study, shown in Figure 3.1, the International Geosphere-Biosphere Program (IGBP) global vegetation classification scheme from the Moderate Resolution Imaging Spectroradiometer (MODIS) Land Cover Type product (MCD12Q1) was used (Friedl et al., 2010). Since this product is provided at 500-m resolution, it was aggregated using nearest neighbor interpolation to the 1-km scale, which is necessary for the use of other MODIS 1-km products discussed below. Urban extents for 2004 and 2009 are used for corresponding years, and the most recent year available (2012) is used for 2014. Although we see little if any variation at the 1-km scale in our urban area from 2004 to 2012, Potere et al. (2009) consider this product as the most accurate for mapping urban extent at the global scale.

3.3.1 Landsat Fr and Urban Classification

Urban classification of the SPMR was done using cloud-free Landsat 7 scenes for March 8, 2004 and March 6, 2009, and a Landsat 8 scene for February 8, 2014. Digital numbers were converted to radiances for red (ρ_{RED}) and near-infrared (ρ_{NIR}) bands and then radiances were converted to surface reflectance according to Chander et al. (2009). The Normalized Difference Vegetation Index (NDVI) was obtained from surface reflectance from the ρ_{RED} and ρ_{NIR} bands given by:

$$NDVI = \frac{\rho_{NIR} - \rho_{RED}}{\rho_{NIR} + \rho_{RED}}. \tag{3.1}$$

The index is a direct indication of vegetation and cover density, where larger values of NDVI usually correspond to higher vegetation productivity. In this study, outliers in the top and bottom 1% of NDVI values were removed.

Fractional vegetation cover (Fr) derived from NDVI was then calculated based on methodology from Carlson & Ripley (1997) by:

$$Fr = \left(\frac{NDVI - NDVI_O}{NDVI_S + NDVI_O}\right)^2 \tag{3.2}$$

where NDVI_S represents saturation or 100% vegetation cover and NDVI_O represents bare soil. Fr values less than 0 were set to 0 and greater than 1 were set to 1. Resulting maps are shown in Figure 3.2. Gaps in the Landsat 7 scenes were not filled, but were omitted from calculations and then filled with zeros for the wavelet analysis described next.

After obtaining Fr, classifications based on quartiles of Fr from the urban extent in Landsat scenes for 2004, 2009 and 2014 were generated. First, the Fr values were decomposed from 30 m resolution to 960 m or approximately 1 km using two dimensional (2D) wavelet multi-resolution analysis (mra). 2D mra has previously been employed in studies of urban morphology (Myint, 2006, 2010; Mouzourides et al., 2013) and can be a powerfool tool for multi-scale modeling (Ching, 2013; Mouzourides et al., 2013). In this study, Haar wavelets were used because they have a compact, square or rectangular shape which can be useful in the detection of edges in urban forms (Bradshaw & Spies, 1992; Mouzourides et al., 2013). The waveslim package in R was used to do the decomposition (Whitcher, 2014).

Figure 3.3 illustrates the results of decomposing the February 8, 2014 Landsat 8 scene from 30 m to scales corresponding to 120 m, 240 m, 480 m and 960 m. Part of this study investigates the scale-wise change in landscape metrics and Fr diversity and variance, further described below. For the use of the decomposed Landsat vegetation data with MODIS 1-km products, the Fr data

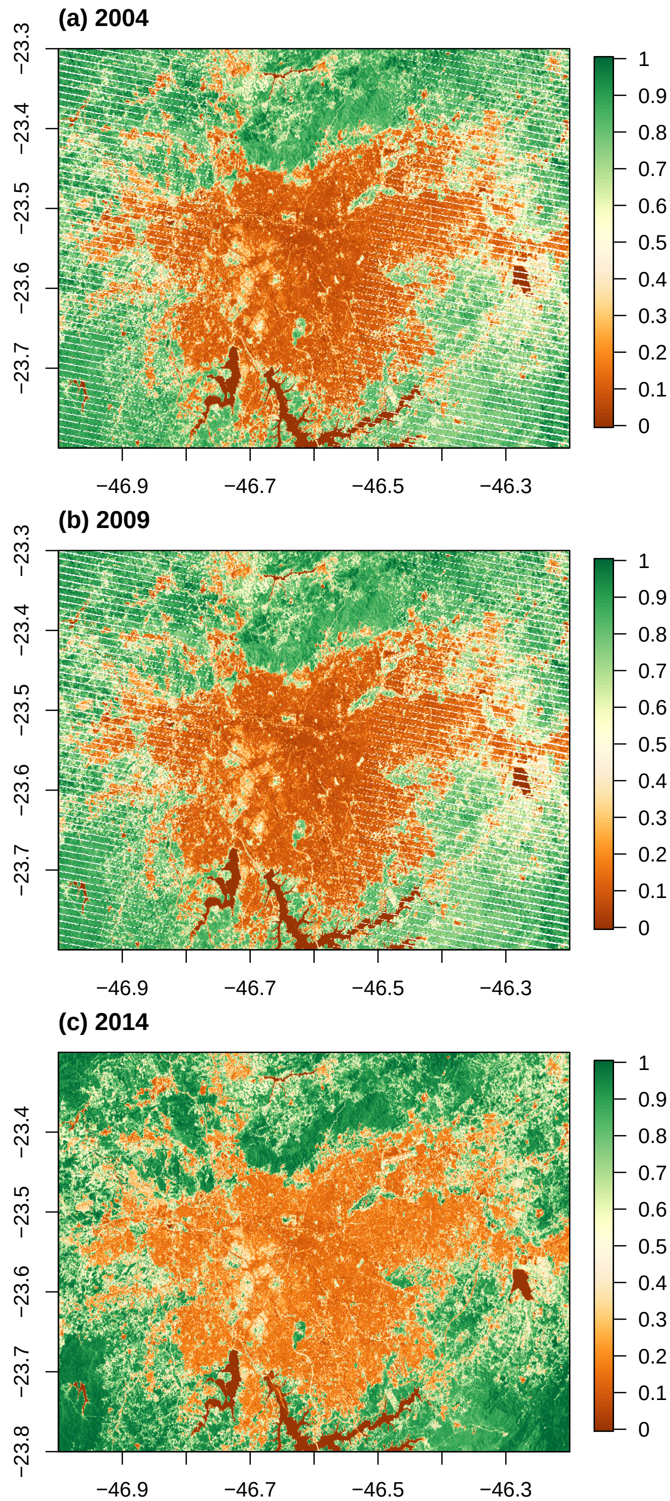

Figure 3.2: Landsat Fr calculated from cloud-free scenes on (a) March 8, 2004, (b) March 6, 2009, and (c) February 8, 2014.

is reprojected to a 1-km resolution. To assess the significance of landscape metrics across scales, landscape metrics were calculated from 1600 bootstrapping samples for each class at each scale assuming a uniform distribution. Upper and lower bounds (95% and 5% confidence intervals) for metrics from bootstrapping samples were then obtained.

The urban classification in this study is comprised of quartiles of Landsat Fr at the 1-km scale for 2004, 2009, and 2014 scenes that correspond to vegetation cover (VC) categories. Values from the minimum Fr to the 1st quartile are classified as VC1, values between the 1st and 2nd quartile are VC2, values between the 2nd and 3rd quartile are VC3, and values from VC3 to the maximum Fr are VC4.

Although this classification method may be simplistic compared to more complex classifications that incorporate variables like building heights and anthropogenic heat fluxes, our purpose is merely a quick generalization of urban forms associated with vegetation distribution that can be consistently determined across years. Selecting quartiles accounts for climatic variation in Fr that would otherwise skew classes from year to year if a fixed threshold was set. However, one should proceed cautiously in making assumptions as to urban density, building height, population density, etc. based solely on vegetation density. For example, parking lots and tall building districts with very low vegetation may be incorporated in the same class according to this classification approach.

To justify the simple classification in this study, we consider that the LCZ classification of Stewart & Oke (2012) was based on criteria from Grigg (1965) that a successful classification system should meet: 1) simple and logical nomenclature, 2) facilitate information transfer through object association with real world, generic classes, and 3) provide for inductive generalization. Like the LCZ classification, fractional vegetation cover is a classification that may allow for global application. However, the purpose of this paper is not to promote this classification, but rather to use it as a surrogate for the forthcoming LCZ classification or any other global urban classification system.

Prior to selecting this method of urban classification, existing classifications at the 1-km scale

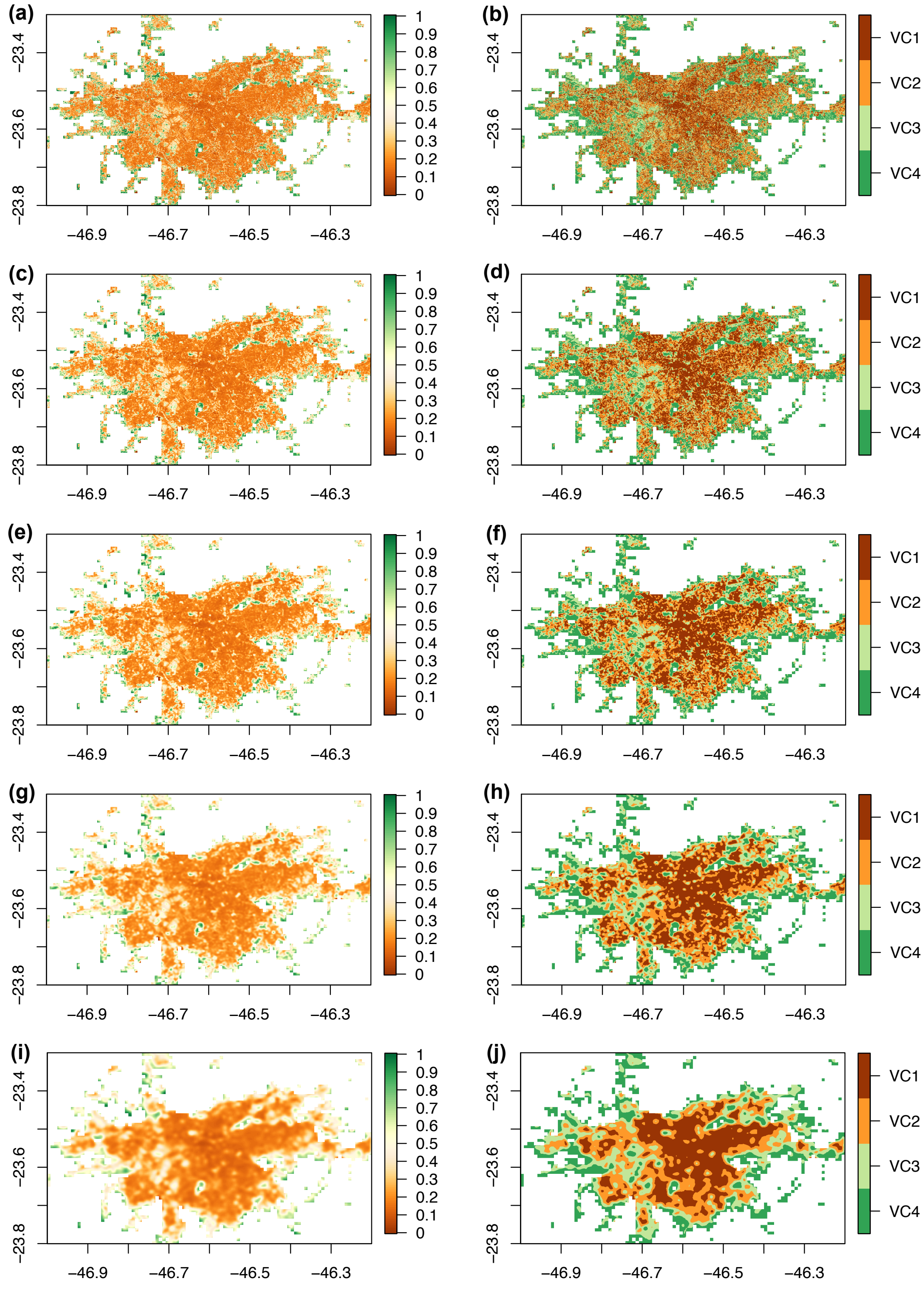

Figure 3.3: From the February 8, 2014 Landsat 8 scene, Fr is seen in the left column and the resulting vegetation cover (VC) classification from quartiles of Fr is seen in the right column at (a) & (b) 30 m and for scales of decomposition at (c) & (d) 120 m, (e) & (f) 240 m, (g) & (h) 480 m, and (i) & (j) 960 m.

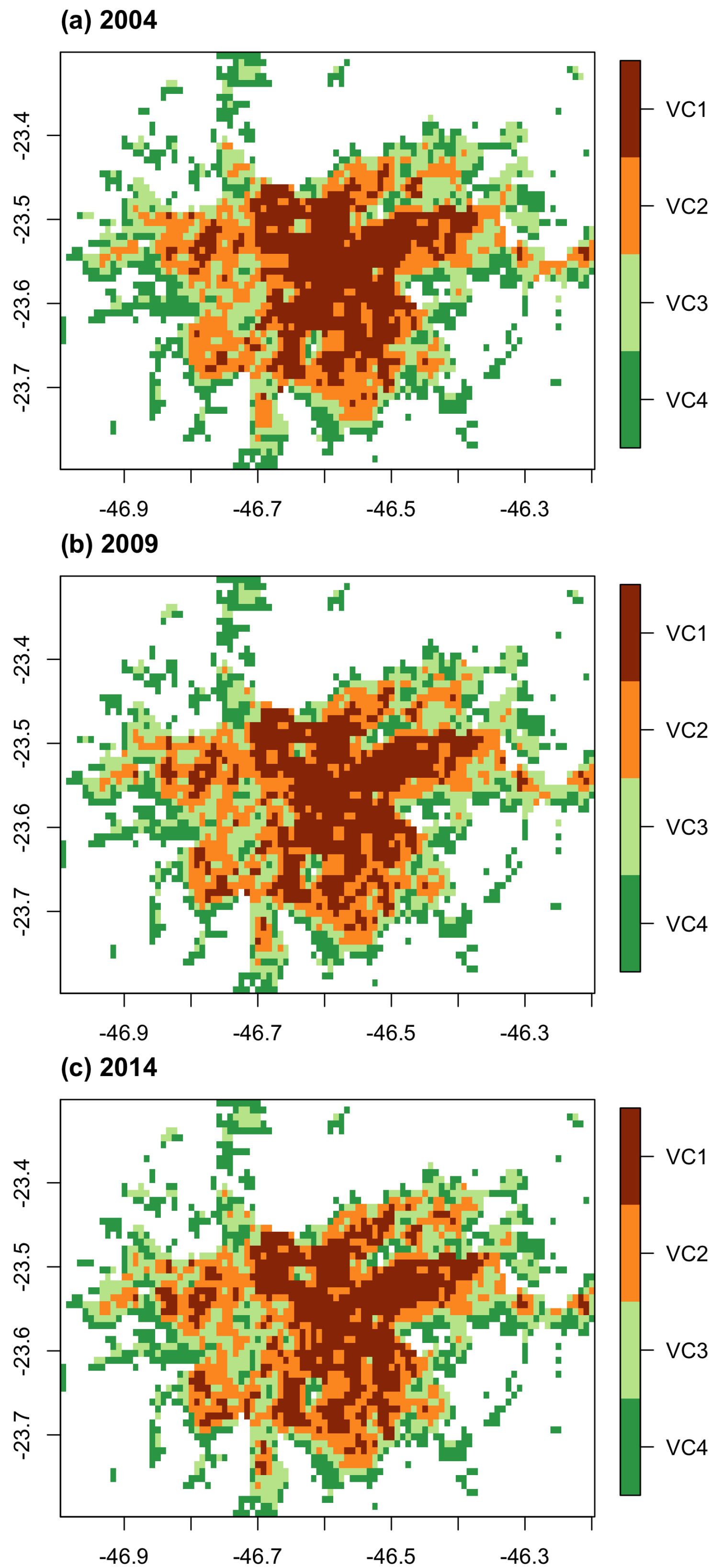

Figure 3.4: VC classification from Landsat Fr decomposed to 960 m with 2D mra and reprojected to the 1-km grid for (a) 2004, (b) 2009, and (c) 2014.

were investigated. The urban classification developed by Jackson et al. (2010), based on Landscan 2004 population densities, building heights, and vegetation fraction thresholds, did not agree with the Fr found in MODIS scenes for the SPMR. For example, based on 5-25% vegetation thresholds for Jackson's high density (HD) category, the actual values of Fr from 2004 MODIS pixels corresponding to that category range from 0 to 86% with a mean of 5.9%. In the medium density (MD) category, the Jackson threshold for vegetation is 20-60% and the corresponding pixel values of MODIS Fr range from 0 to 92% with a mean of 20.5%. Another 1-km global urban classification of impervious surface area based on nighttime lights and population count (Elvidge et al., 2007) was considered, but visual examination between maps showed obvious differences with vegetation cover in Landsat and MODIS imagery for SPMR. Perhaps these discrepancies are due to the skewing of classifications that can occur if vegetation thresholds are set based on urban areas in North America versus Northern Africa, for example, as well as the ambiguity in using nighttime lights to infer urban intensity.

3.3.2 MODIS Fr and LST

Gridded level 3, version 5 MODIS products at the 1-km scale for NDVI (MOD13A3) and LST (MOD11A1) from the Terra platform were acquired from the US Geological Survey (USGS) Land Processes Distributed Active Archive Center (LPDAAC). Composite values of NDVI corresponding to monthly intervals were used, while daily values of LST were obtained. All data were examined for years 2004, 2009, and 2014.

Monthly NDVI values for January, February and March in each year of study were stacked in the raster package in R (Hijmans, 2015) and the maximum NDVI value for each pixel in the region of study was selected to create the maps in Figure 3.6. NDVI values were then converted to Fr based on methods used for Landsat scenes described above.

Taking into account the impact of composite periods and seasonality (Hu & Brunsell, 2013), daily LST values for daytime and nighttime were selected and averaged monthly for 2004, 2009, and 2014. Similar to NDVI, the maximum LST values for daytime and nighttime scenes were

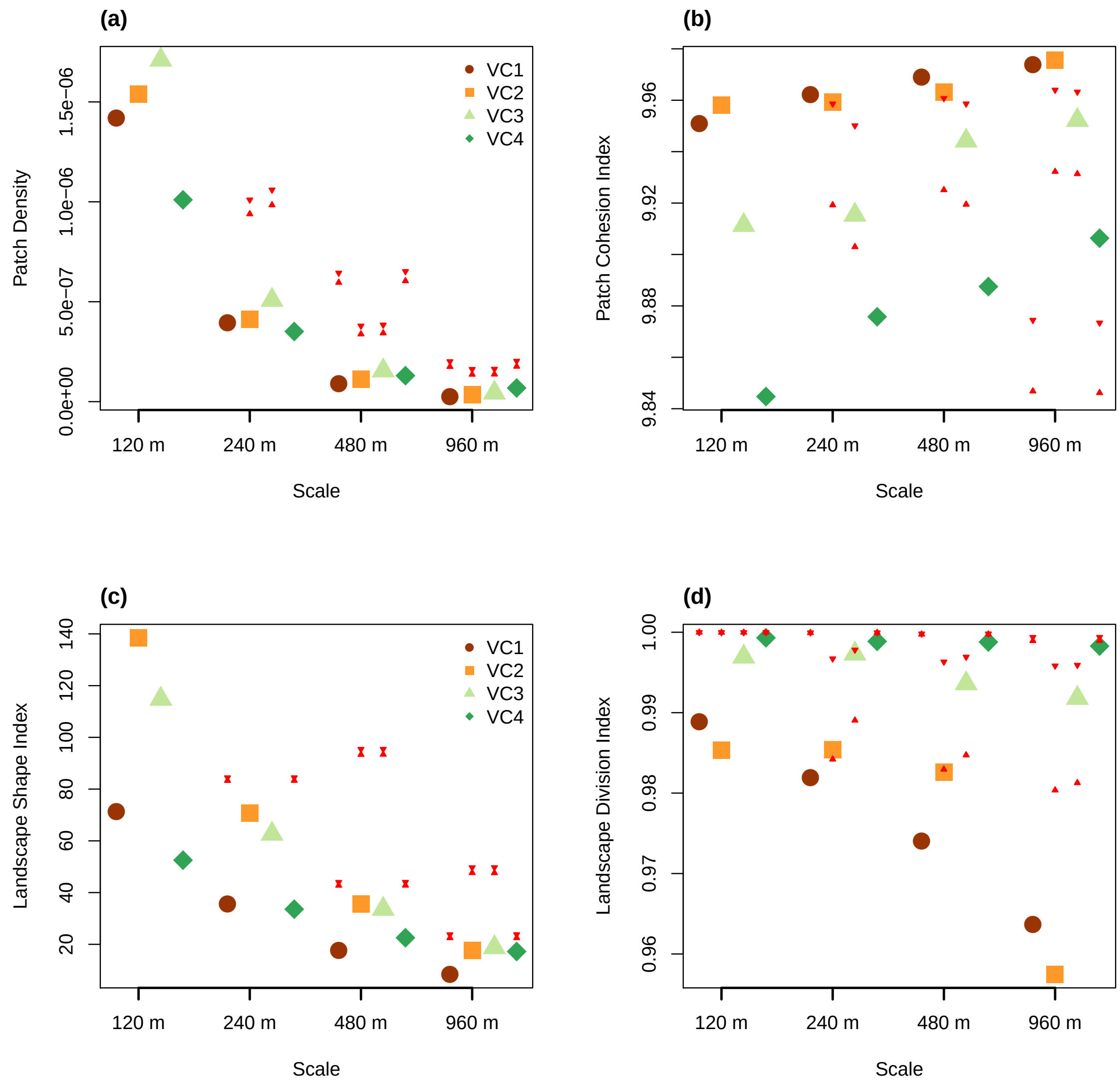

Figure 3.5: Scale-wise change in landscape metrics shown in Table 1 for (a) patch density, (b) patch cohesion index, (c) landscape shape index, and (d) landscape division index. Upper and lower bounds (95% and 5% confidence intervals) for random values of metrics are shown as an upside-down and right-side-up red triangles, respectively. If no red triangles are seen on a plot for a particular class at a particular scale, the plotted value is not within the random value bounds.

Table 3.1: Landscape metrics of VC classifications and Shannon diversity and variance of Landsat Fr within classifications for the original image at 30 m and for scales of decomposition at 120 m, 240 m, 480 m, and 960 m.

	Class	NP	PD	PCI	LSI	LDI	L Fr Mn	L Fr SDI	L Fr σ^2
960 m	VC1	47	2.55E-08	9.97393	8.37727	0.96365	0.18770	13.2181	0.00026
	VC2	63	3.42E-08	9.97568	17.68998	0.95746	0.2440	13.21818	0.00045
	VC3	81	4.39E-08	9.95224	18.80094	0.99180	0.3522	13.21443	0.00187
	VC4	126	6.84E-08	9.90633	17.22663	0.99828	0.5595	13.20881	0.00851
480 m	VC1	168	9.12E-08	9.96910	17.70679	0.97402	0.17690	13.217	0.00029
	VC2	204	1.11E-07	9.96314	35.58440	0.98260	0.2307	13.21827	0.00039
	VC3	285	1.55E-07	9.94420	33.62744	0.99362	0.3402	13.21299	0.00208
	VC4	239	1.30E-07	9.88751	22.54338	0.99878	0.5850	13.20429	0.01245
240 m	VC1	728	3.95E-07	9.96222	35.61264	0.98191	0.16560	13.21486	0.00033
	VC2	761	4.13E-07	9.95930	70.62878	0.98539	0.2180	13.21819	0.00036
	VC3	936	5.08E-07	9.91537	62.65367	0.99732	0.3300	13.21058	0.00249
	VC4	647	3.51E-07	9.87577	33.58574	0.99886	0.6161	13.20013	0.01689
120 m	VC1	2618	1.42E-06	9.95104	71.23806	0.98887	0.15390	13.21155	0.00039
	VC2	2844	1.54E-06	9.95805	138.48554	0.98534	0.2059	13.21788	0.00035
	VC3	3149	1.71E-06	9.91138	114.80833	0.99694	0.3209	13.20808	0.00289
	VC4	1858	1.01E-06	9.84473	52.53598	0.99931	0.6482	13.19751	0.02069
30 m	VC1	19337	1.05E-05	9.92143	199.30087	0.99520	0.1496	13.17793	0.00307
	VC2	32712	1.77E-05	9.67441	330.29657	0.99982	0.1859	13.21725	0.00033
	VC3	25175	1.37E-05	9.72291	286.06859	0.99970	0.3099	13.20162	0.00398
	VC4	11456	6.22E-06	9.78313	116.59367	0.99967	0.6979	13.19396	0.02365

selected for each pixel from monthly values in each year of study. Both MODIS Fr and LST were reprojected from a Sinusoidal to a geographic projection with WGS84 datum and resampled to 1 km.

3.3.3 Landscape Metrics

Landscape metrics (LM) are static and discrete measures of landscape configuration that have been broadly applied to quantify landscape patterns and structures, particularly in the fields of biogeography and landscape ecology (Bolliger et al., 2007; Turner, 2005). In the last decade, LM have been increasingly used to evaluate urban land cover/use change (Herold et al., 2002; Uuemaa et al., 2013; Herold et al., 2005). Although LM may be helpful in monitoring changes in

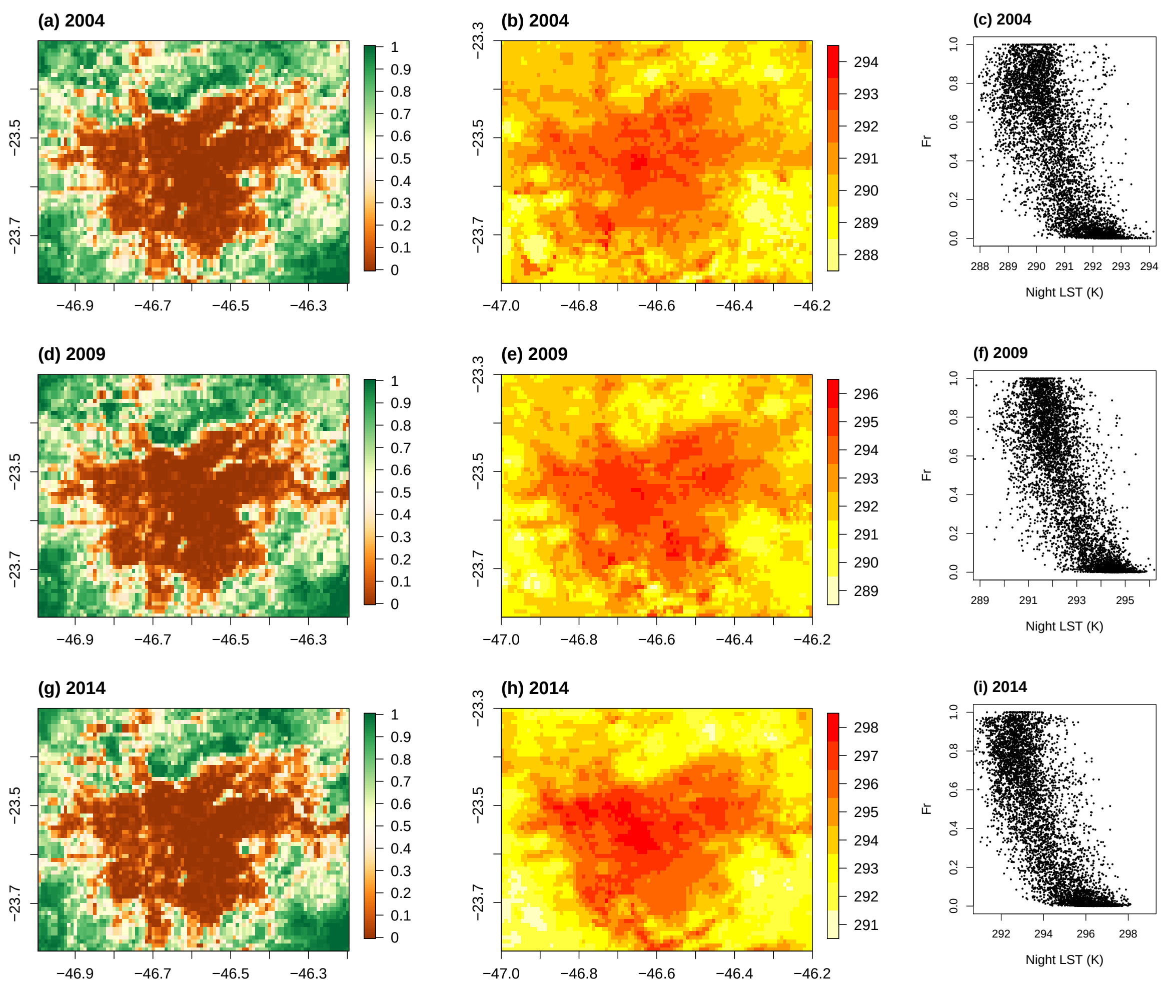

Figure 3.6: Maximum NDVI from monthly product during the growing season (Jan, Feb, Mar) in (a) 2004, (d) 2009, and (g) 2014. Maximum nighttime LST for all months in (b) 2004, (e) 2009, and (h) 2014. Scatterplot of LST and NDVI values for (c) 2004, (f) 2009, and (i) 2014.

Table 3.2: Minimum, maximum and quartile values of Landsat Fr per year for VC classifications.

Fr	**2014**	**2009**	**2004**
Min	0.12	0.07	0.08
Qu1	0.21	0.15	0.17
Qu2	0.29	0.23	0.25
Qu3	0.43	0.37	0.39
Max	0.9	0.82	0.76

landscape patterns, difficulties in relating them to underlying biophysical or ecological processes remain (Kupfer, 2012; Li & Wu, 2004; Turner, 2001). Attributing changes in LM to processes in urban landscapes, where innumerable socio-economic factors are at play, may prove to be even more difficult (Herold et al., 2005).

The inability to compare LM across scales has been one of the main difficulties encountered in progressing from landscape pattern to process. However, Wu et al. (2002) conducted a multi-scale analysis of landscape heterogeneity and found that some landscape metrics respond predictably over a range of grain sizes. These include metrics for the number of patches, patch density, landscape shape index, and patch size coefficient of variation. Landscape metrics that respond less predictably include the patch diversity or Shannon diversity index, and ones that respond erratically include landscape fractal dimension (Wu et al., 2002).

Once urban classifications based on Landsat Fr were created and MODIS data were used to produce reprojected and resampled scenes of maximum Fr and nighttime and daytime LST as described above, landscape metrics were applied to each urban class and year of study. Based on previous studies of correlations and scale issues (Wu et al., 2002), four landscape metrics were selected and applied to the VC classifications using FRAGSTATS (McGarigal et al., 2002) functions in the SDMTools package in R (VanDerWal et al., 2014). Patch density (PD) as a measure of density is defined by:

$$PD = \left(\frac{n_i}{A}\right) \tag{3.3}$$

where n_i is the number of patches in the landscape of a certain class and A is the total landscape area in m^2. The patch cohesion index (PCI) as a measure of connectivity is:

$$PCI = \left[1 - \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} p_{ij}^{*}}{\sum_{i=1}^{m}\sum_{j=1}^{n} p_{ij}^{*}\sqrt{a_{ij}^{*}}}\right] \cdot \left[1 - \frac{1}{\sqrt{Z}}\right]^{-1} \cdot (100) \tag{3.4}$$

where p_{ij}^{*} is the perimeter of patch ij in terms of number of cell surfaces, a_{ij}^{*} is the area of patch ij in terms of number of cells, and Z is the total number of cells in the landscape.

The landscape shape index (LSI) is a standardized measure of the total edge in the landscape, which can indicate landscape disaggregation or geometric complexity:

$$LSI = \frac{0.25E^{*}}{\sqrt{A}} \tag{3.5}$$

where E^{*} is the total length (m) of the edge, including the entire landscape boundary, and A is again the total landscape area in m^2. The landscape division index (LDI) is a measure of aggregation and can be calculated by:

$$LDI = \left[1 - \sum_{i=1}^{m}\sum_{j=1}^{n}\left(\frac{a_{ij}}{A}\right)^{2}\right] \tag{3.6}$$

where a_{ij} is the area in m^2 of patch ij and A is the total landscape area in m^2.

The Shannon diversity index (SDI) from the vegan package in R (Oksanen et al., 2015) was also applied to measure diversity within and between classes and years for Landsat Fr, MODIS Fr and MODIS LST. The variance (σ^2) of Landsat Fr was calculated per class and scale of decomposition.

Distributions of MODIS Fr and LST in each VC class in 2004, 2009, and 2014 were examined. Fr accounts for a measure of greenness, important to the determination of increased environmental sustainability in urban areas (Jabareen, 2006). The difference between maximum daytime and nighttime LST were also computed to investigate the change in urban temperatures within each class. This change gives us an indication of the impact of the urban heat island effect in SPMR and how it may be influenced by urban form over time.

Table 3.3: Landscape metrics applied to VC classifications and Shannon diversity index (SDI) applied to values of MODIS Fr and MODIS LST within VC classifications for 2004, 2009 and 2014.

	Class	NP	PD	PCI	LSI	LDI	Fr Mn	Fr SDI	LST Mn	LST SDI
2004	VC1	22	1.12E-08	9.37115	5.66667	0.94989	0.01357	5.57912	311.2	6.450469
	VC2	52	2.64E-08	8.80783	11.60784	0.98929	0.04992	5.98417	308.8	6.448887
	VC3	73	3.71E-08	8.11575	13.60784	0.99648	0.1605	6.17802	306.1	6.450467
	VC4	93	4.72E-08	7.77327	13.56863	0.99776	0.3899	6.32829	303.4	6.450467
2009	VC1	28	1.42E-08	9.34593	6.74510	0.95537	0.009794	5.48675	312.5	6.448888
	VC2	57	2.90E-08	8.86379	12.41176	0.98851	0.0385	5.96433	310.4	6.447304
	VC3	75	3.82E-08	8.12572	13.70588	0.99653	0.1463	6.17227	307.7	6.448886
	VC4	97	4.93E-08	7.46366	13.88235	0.99836	0.3793	6.31891	304.8	6.448884
2014	VC1	27	1.37E-08	9.30387	6.96078	0.96105	0.01184	5.35236	312.7	6.448887
	VC2	59	3.00E-08	8.60515	12.86275	0.99315	0.03785	5.92196	311.3	6.447303
	VC3	75	3.82E-08	8.08953	14.07843	0.99657	0.1326	6.14257	309.2	6.448885
	VC4	101	5.14E-08	7.49037	14.37255	0.99830	0.3562	6.30964	306.5	6.448884

3.4 Results

3.4.1 Spatial Variability in 2014

The decomposition of Landsat Fr from the original 30 m resolution to the 960 m scale and the VC classification from quartiles at each scale results in changes in landscape metrics, some predictable and some not (Table 3.1 and Figure 3.5). Similar to the findings of Wu et al. (2002), number of patches, patch density, and landscape shape index decrease predictably from 30 m to 960 m. On the other hand, the patch cohesion index increases in an unpredictable pattern from 30 m to 960 m, and the landscape division index decreases in an unpredictable pattern. In these last two metrics (Figure 3.5 b & d), VC1 seems to increase or decrease in more predictable patterns than VC2, VC3, and VC4. Also, the significance of VC2 and VC3 values in both metrics starts to break down after the 120 m scale, though VC2 values are significant again at the 960 m scale. Therefore, if we want to examine landscape metrics at the 1-km scale, yet be able to predictably relate our findings to finer scales, we can feel confident in using the patch density and landscape shape index. In areas with very little vegetation corresponding to class VC1, for example, we may also be able to examine the results of patch cohesion index and landscape division index.

The impact of scale on the diversity and variance of Fr within each class is seen in the last two columns of Table 3.1. Although Wu et al. (2002) have identified Shannon's Diversity Index as a metric that does not follow a simple scaling function and, therefore, is difficult to predict across scales, we consider the information it provides us between classes within each scale to be useful (Brunsell & Young, 2008; Brunsell & Anderson, 2011). Across classes at each scale of decomposition, SDI remains the highest in VC2. Landsat Fr variance decreases from 30 m to 960 m, except for Fr in VC2 where the variance increases from the finest to coarsest scale. These findings highlight the need to further explore what contributes to the uniqueness of class VC2 as well as the differences in scaling relations that may exist in other urban classifications.

3.4.2 Temporal Variability at the 1-km Scale

When we examine landscape metrics in classes at the 1-km scale across years (Table 3.3), we consider that patch density and landscape shape index have a predictable, inverse power relationship (or decreasing power law relationship) from the 30-m to 1-km scale, which is supported by Wu et al. (2002). Given these scaling relations and the significance of these metrics in all classes (Figure 3.5), we see that patch density, or the number of patches on a per unit area basis, is increasing in all classes from 2004-2014 (Table 3.3). Within each year, patch density increases with increasing vegetation or mean MODIS Fr from classes VC1 to VC4. The landscape shape index, a standardized measure of total edge adjusted by landscape size, also increases from VC1 to VC4 in each year and in all classes from 2004-2014. Landscape shape index can be interpreted as a measure of patch dispersal, which also gives us an indication of the overall geometric complexity of the landscape. In other words, vegetation is more dispersed in classes with greater Fr and disaggregation in vegetation fraction is growing. So, while density is increasing in patches of all classes in SPMR over the decade, there is a general trend toward increasing complexity or mixed land uses.

Based on our results from the spatial trends of landscape metrics above, our results for patch cohesion index and landscape division index may not be significant for classes VC2, VC3 and VC4 at the 1-km scale. Therefore, we will discuss only the changes in VC1, the class with the least amount of vegetation fraction or the most amount of impervious surface area. For this class, the patch cohesion index indicates that the physical connectedness within patch type is decreasing from 2004 to 2014. At the same time, the landscape division index is increasing in class VC1 from 2004 to 2014. The closer the landscape division index is to 1, the greater the probability that two randomly chosen pixels in the landscape are not in the same patch, or the greater the division. Overall, both of this indices are indicating that the area of SPMR with the least amount of vegetation is becoming less connected or more divided.

As expected, the Shannon diversity index for MODIS Fr within each year increases from areas with less vegetation (VC1) to areas with more vegetation (VC4) in the SPMR. Excluding the VC2 class, the SDI for maximum nighttime LST increases from VC4 to VC1 for each year, suggesting there is consistently higher richness in LST in areas with less vegetation than more. This excludes the VC2 class which is consistently lower in all years than all other classes, indicting again the added difficulty in assessing biophysical sustainability in these particular parts of the city.

Looking at Figure 3.7, we see that the distribution of daytime and nighttime LST shifts downward from VC1 to VC4 classes. However, over the years the distributions of both daytime and nighttime LST are shifting upwards in each class. From 2004-2014, the difference in daytime LST to nighttime LST can be seen decreasing in Figure 3.8a-c. The change in temperature is less in VC3 and VC4 and higher in VC2 and VC1 classes. This decrease in temperature difference between the day and night in less vegetated classes over the last decade could indicate either an increase in night temperatures or a decrease in day temperatures or both depending on the pixel and year considered. Given the temperature distributions seen in Figure 3.7b, where nighttime temperatures are rising faster than daytime temperatures in VC1, we can infer that heat dissipation from the urban boundary is occurring less at night lowering the difference between maximum daytime and nighttime temperatures.

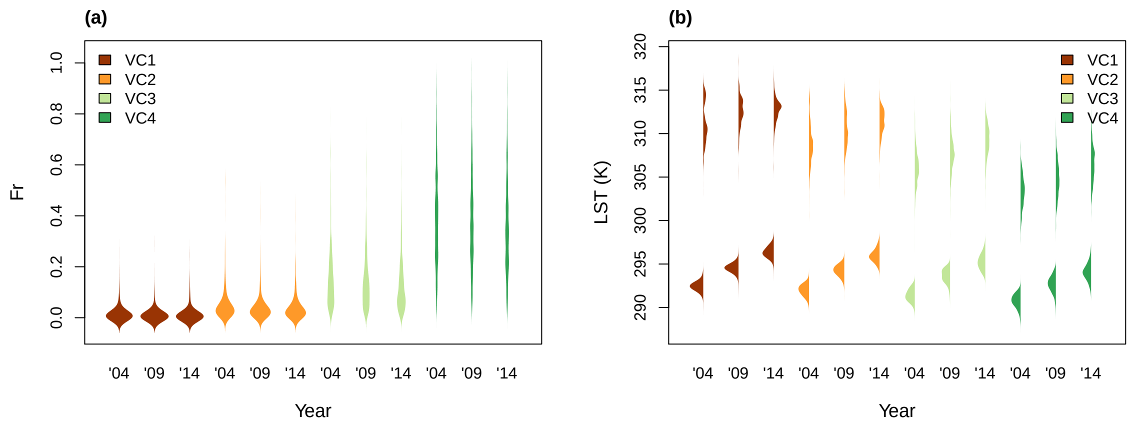

Figure 3.7: Distributions by VC class and year for MODIS (a) Fr and (b) maximum daytime and maximum nighttime LST. The VC classes were determined by Landsat Fr distributions in each year.

3.5 Discussion

Considering our definition of more sustainable urban forms and our results using landscape metrics, Fr, and LST, we can now evaluate the urban region of SPMR from 2004 to 2014 at the 1-km scale and see how each metric relates to biophysical sustainability. As outlined by Jabareen (2006), more sustainable urban forms may translate to high density, diversity, complexity (mixed land use), compactness, connectedness (sustainable transportation), and greening at varying scales. These characteristics are echoed in urban planning concepts of transect design and biophilic urbanism (Duany & Talen, 2007; Beatley, 2009), however they may not sufficiently consider the urban heat island effect, its interaction with global climate change, and its impact on urban sustainability. Therefore, our definition of more sustainable urban forms requires that these urban form characteristics help mitigate the urban heat island.

To examine changes of urban form in SPMR from 2004 to 2014, reliable metrics include patch density as a measure of density, landscape shape index as a measure of complexity (mixed land use), mean MODIS Fr as a measure of greenness, and mean MODIS LST as well as changes in daytime to nighttime LST as proxies for the UHI in SPMR (Table 3.3 and Figure 3.8). Metrics that are shown to be unreliable except in the VC1 class, include the patch cohesion index as a measure

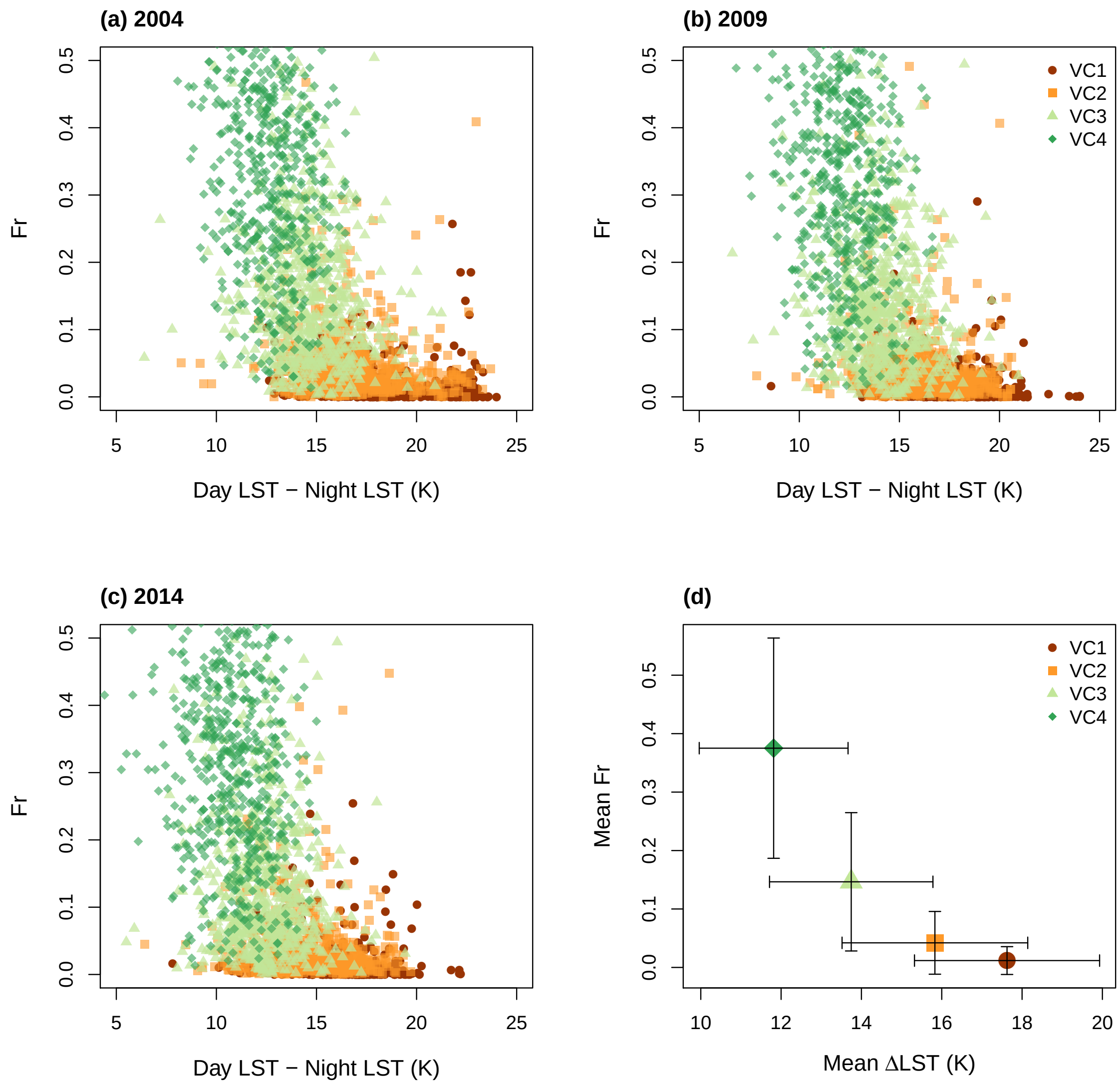

Figure 3.8: Maximum daytime minus maximum nighttime LST in each class for (a) 2004, (b) 2009, and (c) 2014. (d) Mean LST values for all years by class plus or minus the standard deviation (SD) shown as error bars.

of connectivity and the landscape division index as a measure of (dis)aggregation (Figure 3.5).

Since we know landscape metrics are sensitive to both the pixel size and the urban extent (Turner, 2005; Wu et al., 2002), our investigation of changes in landscape metrics from the 30-m to 1-km scale can help us interpret our findings at the coarser scale. Metrics that have an inverse power relationship (or decreasing power law relationship) from the 30 m to 1 km include the patch density and landscape shape index. So, when discussing the increase in urban density and complexity (mixed land use) of SPMR at the 1-km scale from 2004-2014, we acknowledge an underestimation of the value of these metrics in each year in relation to finer scales.

The differences in scaling of metrics in Figure 3.5 show that more landscape metrics may be reliable up to the 120-m scale compared to the 1-km scale. Using fractal dimensions of NDVI and vegetation fraction images, Weng et al. (2004) found that complexity increases with pixel aggregation up to 120 m but then decreases with coarser scales. They also found the strongest relationship between vegetation indices and LST at 120 m. Clearly, this is a scale of importance for evaluating urban forms as it is approximately the scale of a city block (Weng et al., 2004). Beyond this scale is also where metrics for our VC2 classification become unrealiable. One explanation for this occurrence may be the combined low variance in Fr values but high SDI of Fr at the 30 m scale (Figure 3.5), which may correspond to similar types of vegetation but greater variation in land uses and urban canyons in these areas.

At the 1-km scale, we found that urban density, complexity, and temperatures increase from 2004 to 2014 in all VC classes in SPMR. On the other hand, overall greenness decreases in all classes from 2004 to 2014, with the exception that greenness rises slightly in VC1 from 2009 to 2014. In the VC1 class with the least overall amount of vegetation, patch density also decreases from 2009 to 2014, and connectivity decreases or division increases over the decade. It is important to remember that the landscape metrics in this study are directly related to Landsat Fr due to our urban classification scheme, but a class like VC1 with the least amount of vegetation also has the greatest amount of impervious surface area.

Our findings that a higher patch density and landscape shape index is associated with higher

LST agree with results from Li et al. (2011) who found that mean LST was positively correlated with patch density and landscape shape index for residential land. In addition, our findings on the change in temperature (Figure 3.8) across urban classes from 2004-2014 agree with results from numerical surface and atmospheric models in (Stewart & Oke, 2012) where diurnal temperature ranges decrease with increased impervious surface area. Since the current trends in complexity, density and greening in SPMR are associated with increased urban temperatures, characteristics of urban form are not working to mitigate the urban heat island and, therefore, are not sustainable. To move back toward sustainability, modifications to complexity, density, and greenness of urban form may be attempted to reduce urban temperatures.

Using our results from Figure 3.8d, we conceptually frame sustainable urban forms in LST-Fr space in Figure 3.9. In this manner, we can imagine the economic costs in concentric circles needed to move urban forms that fall within the warmer part of the triangle back toward the cooler edge of the triangle. To work toward sustainability, we can increase greening, but we must also work out tradeoffs between complexity and density, which may both contribute to or take away from mitigation and adaptation efforts.

As previously discussed, SPMR is unique in its reliability on renewable energy technologies and hydroelectricity, its relatively high urban density but low per capita income, and being the only one of 12 megacities that has a lower carbon footprint than both the national and international averages (Sovacool & Brown, 2010). Because emissions related to energy use were relatively low, city managers and planners may not have focused enough on implementing energy efficiency in building standards over the last decade. At the same time, their efforts to increase greenspaces and density in certain areas may be evident in our results, in a slight increase in greenness from 2009-2014 and in the decreasing patch density and increasing division found in the least vegetated areas or VC1.

Tree distribution is important, and a thin, distributed canopy offers more shade than a dense cluster, which contributes less to urban cooling (Stone & Rodgers, 2001; Li et al., 2011). Reflecting some of the tradeoffs that may exist in considering vegetation or impervious surface density and

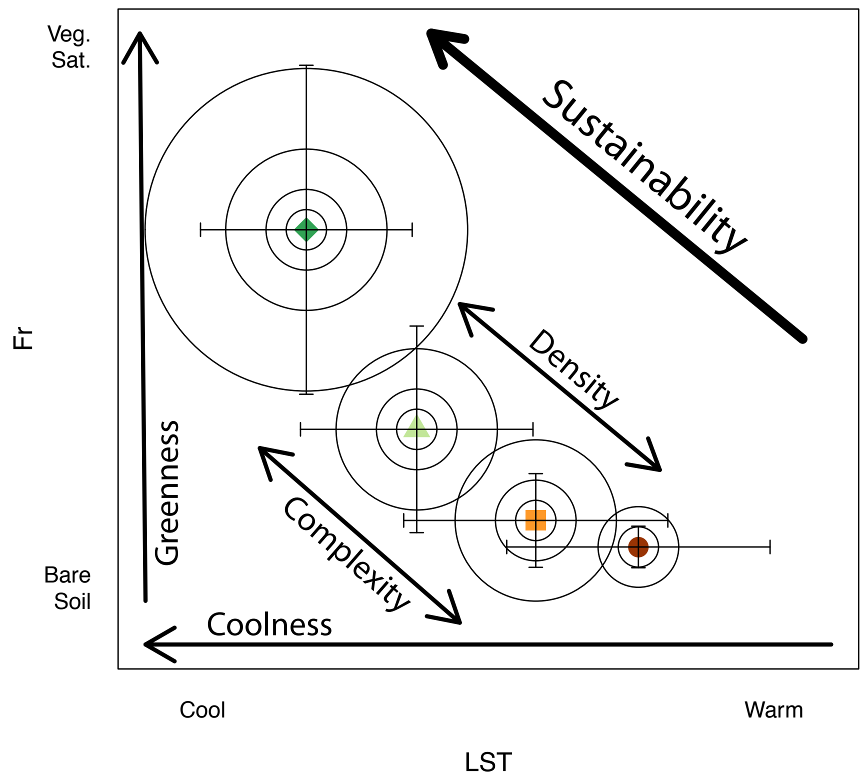

Figure 3.9: Conceptual plot of what the move toward more sustainable urban forms may entail. To move through concentric circles toward greater sustainability may cost a certain amount to work out the tradeoffs between density and complexity and to increase greenness.

complexity shown in Figure 3.9, different tree types are recommended for parks, parking lots, and urban canyons. Tall, narrow trees are optimal for shading canyon walls and sidewalks while reducing downdrafts at the base of buildings (McPherson, 1994). Using the framework of Figure 3.9, we can imagine the trajectory of a pixel from a city or urban area that has minimal greening but undergoes greening efforts. If the greening effort contributes to cooling, we should see the pixel move not only up but to the left, or toward the upper left corner of the triangle, indicating greater sustainability of the associated urban form. At the same time, there may be a tradeoff between cooling associated with evapotranspiration and the water demand and cost for maintaining vegetation. Increased tree cover in warming urban areas may also release more VOCs contributing to elevated ozone and poor air quality (Cardelino & Chameides, 1990).

Field monitoring of an urban canyon, a city square, and a city park in the São Paulo Metropolitan Region (SPMR) resulted in the finding that the park was up to 2°C cooler than the urban canyon or city square (Spangenberg et al., 2008). The same study found that the addition of trees to the urban canyon had a limited cooling effect, mostly on the street surface, and that it lowered wind speeds up to 45%. Therefore, we cannot rely on greening alone as a means for reducing the UHI.

Technologies to provide urban cooling beyond greening are widely known and readily available, though policies and funding to apply them may be lacking (Smith & Levermore, 2008). Thermal efficiency outside and inside current buildings can benefit from albedo modification, appropriate insulation materials, and glazing materials that can reduce heat transmission by 75% while maintaining light infiltration. Other planning techniques are less likely to apply in existing buildings, but should be required in the creation of new built-up areas. These include optimal street orientation for urban ventilation, building orientation for passive solar, and placement of parks and water bodies where advection can result in cooling winds through built up areas. Years ago, Oke (1988) also proposed an ideal aspect ratio (building height/canyon width) of 0.4 to 0.6 and building densities (roof area/total surface area) of 0.2 to 0.4 for mid to high latitude cities.

How density impacts LST and UHI may be most related to canyon geometry or building height and vegetation distribution that impacts the sky view angle, radiation absorption and reflection, and opportunities for shade and ventilation during day and nighttime hours (Li et al., 2011; Oke, 1981; Smith & Levermore, 2008; Stone & Rodgers, 2001). Single family parcels examined in Atlanta, USA showed a significant positive relationship between parcel size and net thermal emissions, which supports the theory that expansive urban form, also known as urban sprawl, contributes more per parcel to radiant heat energy than dense urban form (Stone & Rodgers, 2001). The exposed surface area is considered key to thermal emissions and supported by further research in Shanghai, China where residential land use with lower buildings and vegetation cover had higher LST than areas with high buildings or areas with high vegetation cover (Li et al., 2011). In this study, LST was calculated from Landsat scenes, which did not allow for the consideration of nighttime temperatures. Urban areas with high density, tall buildings can create temporary cooling islands during the day due to shadows. However, at night urban canyons trap long-wave emissions that are absorbed in building materials and reflected between each other delaying urban cooling. Moderate urban density that allows for dispersed greening and provision of shade while still maintaining ventilation may best satisfy both mitigation and adaptation strategies (Hamin & Gurran, 2009).

Contrary to other large cities in South America where there are falling density trends and increasing urban sprawl (Inostroza et al., 2013), our results indicate that SPMR is increasing in density overall. Whereas, an increase in urban density may be considered characteristic of more sustainable form in relation to transportation, we speculate that increased density at least partially contributes to observed increases in urban temperatures in SPMR. However, influences of increased global temperatures must first be ruled out.

Sources of error in our metrics and classification method may be introduced by the different timing of the Landsat scenes, as well as the Landsat 7 gaps. Shao & Wu (2008, pg. 507) address the error that is carried over from classified thematic maps and remote sensing products to landscape analysis and report that "a high degree of classification accuracy is required for assuring the consistency and reliability of landscape metrics." Environmental factors may also impact remote sensing data, such as the finding that humidity in cities increases the variability of MODIS temperatures (Hu & Brunsell, 2015). Finally, the spatial configuration of the UHI can be influenced by seasonal changes, so the use of annual maximum daytime and nighttime LST in this study does not address these seasonal variations.

Despite the uncertainty in error propagation throughout this study, there are some clear tradeoffs and advantages for assessing more sustainable urban forms at the 1-km scale. Evaluating and monitoring the sustainability of global urban forms over time may be more attainable with high frequency satellite data like MODIS. Together with finer scale classifications (i.e. LCZs), broader scale temporal analyses of how urban forms contribute more or less to UHI will help policymakers and urban planners regulate and monitor biophysical sustainability, while contributing to social and economic sustainability.

3.6 Conclusions

In this study, we evaluate the possibility of using MODIS products and landscape metrics to assess more sustainable urban forms at the global scale. Our case study of SPMR, a city that exhibits relatively low carbon emissions and high urban density, suggests that studies of sustainable urban form must quantify the influence of forms on the mitigation of the urban heat island. Urban complexity, density and temperatures increase from 2004 to 2014 in SPMR, while urban greenness decreases in most urban classes. From 2009-2014, policies associated with efforts for urban greening may be evident in the VC1 class, which has the least amount of vegetation or greatest impervious surface area. However, this effort seems to have little if any effect on urban cooling.

Across scales, landscape metrics of patch density and landscape shape index provide reliable and predictable quantifications of urban density and urban complexity or mixed land uses. Tradeoffs between urban complexity and density exist in relation to urban heat islands, and figuring out how these tradeoffs relate to LST-Fr space around the globe may be the next step in defining sustainable urban forms.

Acknowledgements

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Chapter 4

Indigenous Ecological Calendars Define Scales for Climate Change and Sustainability Assessments

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Abstract

Identifying appropriate temporal and spatial boundaries for assessments of human-environment systems continues to be a challenge in sustainability science. The livelihoods of Indigenous peoples in the northwestern Brazilian Amazon are characterized by complex ecological management systems entwined with sociocultural practices and sophisticated astronomical and ecological calendars. Sustainability of fisheries and bitter manioc production, key elements of food systems and economic activities in this region, depend on cyclic high river levels for fish spawning as well as periods of dry days for preparation of agricultural fields. Since 2005, participatory research has been underway between Indigenous communities of the Tiquié River and the Brazilian Socio-environmental Institute (ISA). Indigenous agents of environmental management (AIMAs) keep notebooks of ethno-astronomical, ecological, and socio-economic observations of the annual cycles, and some of them have reported that river levels and dry periods have become more irregular in some years. To investigate how these possible climatic changes may impact the sustainability of resources, we share knowledge from the Tukano ecological calendar with methodology for examining changes in precipitation and river levels and their interactions at multiple timescales. Our collaboration indicates that high spatial and temporal variability in precipitation patterns and river levels may complicate climate change and sustainability analyses. However, combining results from participatory research with novel methods for climate analysis helps identify a four-day trend in precipitation that may impact agroecosystem management. Indigenous participation in systematic data collection and interpretation of results is essential for distinguishing between socio-economic and climate forcings

and evaluating climate impacts. Continued efforts to bridge Indigenous and Western knowledge systems are vital for sustainable environmental management in Indigenous territories and other regions where traditional management may be challenged in the context of global climate change.

4.1 Introduction

Providing climate services for sustainable livelihoods in remote Indigenous communities is an emerging justice issue. Indigenous peoples and their knowledge are not referenced in the United Nations Framework Convention on Climate Change (UNFCC) or in the Kyoto Protocol and only marginally mentioned in the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report (Smith & Sharp, 2012). For the creation of the IPCC Fifth Assessment Report, Alexander et al. (2011) explored ways in which Indigenous knowledge (IK) could be incorporated with Western scientific knowledge (WK) into climate change assessments, particularly through links between climate-related narratives and remotely-sensed data. By bridging IK and WK systems in areas with limited instrumental records, they uncovered ways in which global climate change has already impacted human-environment systems, particularly in the northern high latitudes (Alexander et al., 2011). Through the co-production of knowledge, defined as a collaborative process with various knowledge sources to create a systems-oriented understanding of a specified problem (Armitage et al., 2011), an improved comprehension of climate change impacts can be achieved (Nakashima et al., 2012; Riedlinger & Berkes, 2001; Salick & Ross, 2009; Tyler et al., 2007).

In the context of global climate change, the bridging of IK and WK systems is essential for the forward-thinking, justice framework proposed by Whyte (2013), where systems of responsibilities (relational responsibilities that enable livelihoods) respond appropriately to climate change by supporting collective continuance or tribal adaptive capacity. Since climate change threatens collective continuance of Indigenous communities, new systems of responsibilities may emerge or be amended to cope with environmental change and the introduction of new scientific and technological systems. Governments, non-governmental decision-makers and scientists are responsible for sheltering and, when appropriate, amending persisting and emerging systems of responsibility. According to Whyte (2013, pg. 528), "institutions that do not shelter the exercise of the responsibilities leave tribes in positions of greater vulnerability vis–à–vis climate change impacts..."

Climate services provide climate information to decision-makers to "facilitate climate-smart decisions that reduce the impact of climate-related hazards and increase the benefits from benign climate conditions" (Hewitt et al., 2012). McNie (2013) argues that co-production of climate information may increase trust and social capital for "useful" climate services. This can be achieved with early and iterative communication with stakeholders, formal or informal research methods to identify stakeholders' needs, and maintenance of social capital to enable information exchanges (McNie, 2013). Since this type of problem-driven research requires tremendous time, funding and resource allocations, it may be helpful to initiate collaborations with boundary organizations (Guston, 2001) and leverage research that is already underway, especially considering the urgency of climate change mitigation and adaptation efforts.

The Socio-environmental Institute (ISA), a Brazilian non-governmental organization (www.socioambiental.org) that supports Indigenous peoples' rights, has undertaken numerous research projects in the Upper Rio Negro during the last 20 years in partnership with the Federation of Indigenous Organizations of the Rio Negro (http://www.foirn.org.br). Through these collaborations, it has been possible since 2005 to organize a network of researchers and knowledge holders among various Indigenous communities of the remote Tiquié River, focusing on environmental management activities (Cabalzar, 2010). As part of this effort, Indigenous agents of environmental management (AIMAs) have recorded their observations of the annual cycle to document the Tukano ecological calendar. The annual calendar covers cycles of fish, amphibians, birds, mammals, insects, plants, daily work in agriculture, fishing, gathering, and hunting, rituals and festivals, prevention and cure of diseases, diet and behaviors. Daily life in communities is the experience of these cycles and processes that are related to the organization of socio–economic activities and rituals (Cabalzar, 2013).

The case study in this paper attempts to deliver useful climate services for sustaining Indigenous livelihoods by leveraging results from participatory research of the Tukano ecological calendar to identify spatial and temporal scales for evaluating climate change and sustainability. Our main query is: What are appropriate spatial and temporal scales for assessing climate change impacts and sustainability in the Tiquié basin? To answer this question, we consider the benefits of bridging IK on the Tukano ecological calendar with a WK analysis of precipitation and river levels at different scales (Berkes, 2006).

Berkes (2006) discuss the need for investigating issues of scale and knowledge systems together for environmental sustainability. Sustainability science necessitates studying human-environment interactions across multiple temporal and spatial scales through the co-production of knowledge and collaborations between scientists, decision-makers and stakeholders (Clark & Dickson, 2003; Kates, 2011). If our study is successful in identifying relevant scales of importance for monitoring climate change and sustainability through the coexistence of IK and WK, it may contribute to the theory that bridging multiple knowledge systems and scales builds adaptive capacity and reduces climate change impacts and vulnerability (Berkes, 2006).

Appropriate spatial and temporal boundaries for monitoring and assessing climate change and sustainability in human-environment systems are difficult to identify (Mayer, 2008). Furthermore, the decoupling of policy actions from sustainability targets, such as reducing climate change impacts, is often attributed to the discrepancies between spatial and temporal scales and is the cause of an ever-widening sustainability gap (Fischer et al., 2007a). This study endeavors to reduce that gap by bridging aspects of IK and WK systems. Although various Indigenous astronomical and ecological calendars have been documented around the world (Sánchez-Cortés & Chavero, 2011; Lefale, 2003; Petheram et al., 2010), temporal scales for climate and sustainability assessments have yet to be defined with Indigenous calendars.

Indigenous calendars constitute native ways of knowing or IK, which according to LaDuke (1994, pg. 127), is "the culturally and spiritually based way in which Indigenous peoples relate to their ecosystems." Across disciplines, attempted "integration" of IK and WK has resulted in widespread difficulties (Casimirri, 2003; Johnson & Mutron, 2007; Mazzocchi, 2008; Nadasdy, 1999; Raymond et al., 2010) primarily owing to the lack of a common ground for evaluating these distinct knowledge systems. According to Wood (1999), IK systems are often "polyrhetorical, emphasizing multiple shifting and context-specific meanings with overlapping and elastic realities," whereas WK systems are "monorhetical, which privileges objective, ideally mathematical, analytical–reductionist, linear…” realities (Johnson & Mutron, 2007, pg. 123). These differences result in diverse understandings of time and connections between cause and effect, due primarily to the nature-culture divide or the removal of spirituality from Western scientific inquiry and the separation of supernatural, human, and biophysical worlds (Johnson & Mutron, 2007).

IK is usually incorporated into WK research through questionnaires, semi-directive interviews, collaborative field projects, and facilitated workshops (Huntington, 2000). Problematic integration of IK with WK occurs when: 1) IK is used as mere "data" to inform Western resource management, 2) WK is used to validate IK, and 3) IK research is conducted based on WK research questions and methods and collected, translated, and interpreted by Western researchers (Casimirri, 2003). These problems often arise when IK is obtained through social science methods before being incorporated into multidisciplinary, scientific analyses (Huntington, 2000).

Acknowledging that this last point may pertain to our research, we instead aim for coexistence of the two knowledge systems (Ball & Janyst, 2008; Johnson, 1992). In the Upper Rio Negro, the question of knowledge production has been discussed with an effort to: 1) recognize the importance and scope of IK, and 2) promote symmetrical relations between Indigenous and non-Indigenous researchers, respecting the characteristics and skills of each and developing intercultural research programs. Long-term initiatives like this require the identification of common ground for communication, often an arduous task considering the cultural and linguistic differences. To facilitate communication, we focus on the commonalities between IK and WK systems, which include the importance of place or "localness" (Johnson & Mutron, 2007), the shared concept of ecosystems (Lertzman, 2010), and the collection of empirical observations (Roberts, 1996). In addition, to approach non-linear realities of IK, our study selects WK methods that identify non-linear sources of change. Berkes (2013, pg. 27) has found that "the two paradigms can best be considered together by combining knowledge in a collaborative way around a particular topic." By concentrating on the topic of spatial and temporal scale, we attempt to overcome challenges of knowledge "integration" and offer instead the opportunity for IK and WK to coexist. Coexistence follows the recommendations from the WIS2DOM workshop to "allow each paradigm to occupy their own separate intellectual space while building bridges between them for dialogue" (Johnson et al., 2013).

The Western researchers involved in this project have lived or worked in the Upper Rio Negro for over 30 years, cumulatively. In 2013, the AIMAs invited the lead author to a workshop to discuss climate change and the Tukano calendar, which provided context for research engagement (Ball & Janyst, 2008). Permission was granted for a climate analysis of the region and for use of the 2005-2008 results of the participatory research pertaining to the calendar. The addition of climate science perspectives reinitiated a phase of collaborative problem framing and team building, a step referred to as Phase A by Lang et al. (2012). The collaboration between ISA and the AIMAs was already long-term, and AIMAs continued to retain full decision-making authority in relation to IK use, an essential aspect of power relations in the coexistence of IK and WK (Brewer & Kronk Warner, 2015; Nadasdy, 1999). Nevertheless, the non-Indigenous researchers of this project acknowledged their position of power, based on their funding and institutional connections, and continue to make every effort to communicate on even ground and follow ethical research practices as outlined in Ball & Janyst (2008).

To illustrate the iterative mechanisms of the co-production of knowledge, its exchange and use between IK and WK spheres, a gear system is used as an example in Figure 4.1. Hence, this paper comprises two distinct stages of analysis: 1) the receipt of IK by Western researchers through participatory research (Figure 4.1d), and 2) a Western climate analysis that provides new output information (Figure 4.1e) for potential climate services in Indigenous territories. After a brief overview of the study region, the case study's pertinence to Indigenous agroecosystems and fisheries, and the climate of northwestern Amazonia, methodologies used in the participatory research are summarized and new methodologies for joining qualitative and quantitative data and conducting the climate analysis are presented. Results are discussed to better define temporal and spatial scales and improve ongoing climate change and sustainability assessments in the ecoregion of the Tiquié River within the larger Amazon Basin.

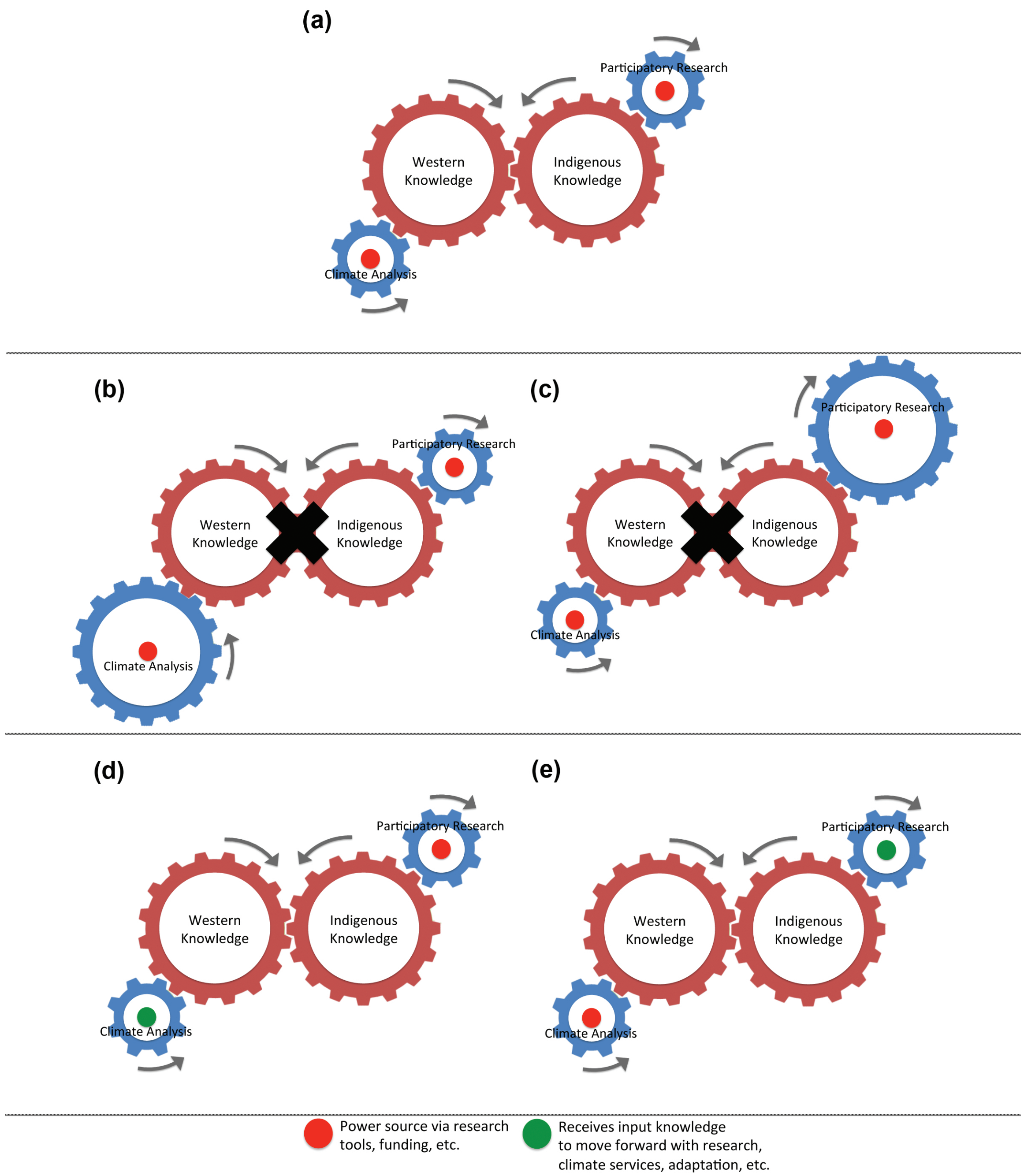

Figure 4.1: (a) As long as both IK and WK gears remain of equal size, weight and speed, determined by the equal size, weight, and speed of the two drive gears (in this case, participatory research and climate analysis) that are powering them, IK and WK can continue to mesh without clashing. (b) and (c) If either of the drive gears gets too big and overpowers the system it will cause IK and WK gears to clash and break, unless the bigger gear slows down or the smaller gear speeds up. Either way, as long as there are two drive gears, there will be no output production of knowledge to propel the system forward. (d) and (e) In order to move forward, either the participatory research or the climate analysis must take turns being the power source in an iterative cycle to provide the other with output knowledge.

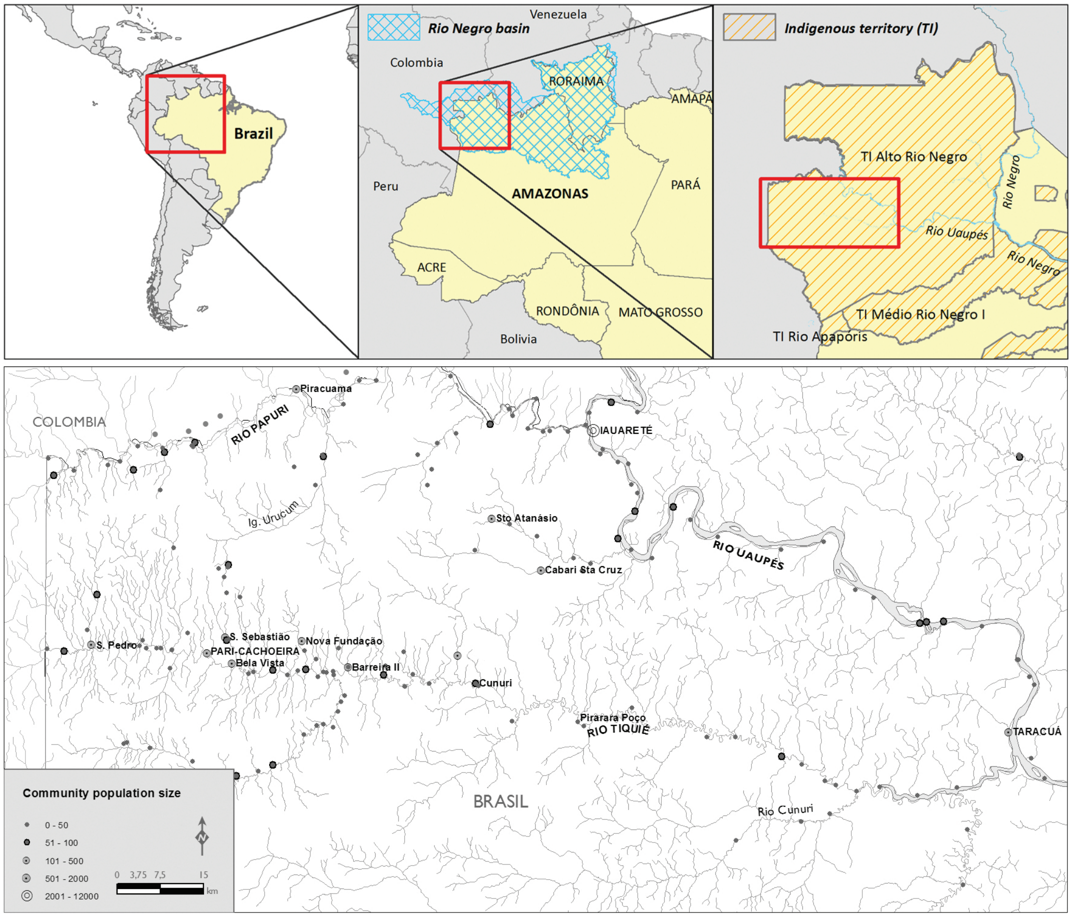

Figure 4.2: Region of study in the state of Amazonas, Brazil and within the Indigenous Territory (TI) of the Alto Rio Negro. Close up of the Tiquié River and Uaupés River with communities by size (courtesy ISA).

4.1.1 Study Region

The Tiquié River is located in the upper, northwestern part of the Rio Negro basin in Brazil and Colombia (Figure 4.2). Its headwaters are in Colombia, and it traverses the western extent of the Guiana Shield draining an area of 5700 km^2. The river runs 374 km of which 321 km are within the Indigenous Territory of the Upper Rio Negro (Terra Indígena (TI) Alto Rio Negro) in the state of Amazonas, Brazil. The Tiquié is an affluent of the Uaupés River, which drains into the Rio Negro, the largest tributary of the Amazon River.

Currently, 21% of the Amazon region is occupied and recognized for exclusive use by Indigenous peoples. In Brazil this area is 22% of federal lands, while in Colombia it exceeds 50%

(RAISG, 2012). The communities of Pari–Cachoeira and Pirarara Poço along the Tiquié River and Taracuá at the junction of the Tiquié River with the Uaupés River parallel the equator from west to east and are located within the TI Alto Rio Negro, federally recognized as an Indigenous territory in 1998. With an area of 8 million ha, it is home to approximately 26,000 people from 20 major ethnic groups. The population of the Tiquié is estimated to be about 4,000, most of whom come from two linguistic groups: the Tukano Oriental, including the Tukano, Desana, Tuyuka and Miriti-tapuya, and the Maku, namely the Hupda and Yuhupda. The language of the Tukano, whose population is prevalent, is used as a *lingua franca* throughout the Tiquié basin. In this paper, words in Portuguese are denoted in lower case italics, while words in Tukano are capitalized.

4.1.2 Case Study

While anthropogenic climate change and associated increased temperatures are expected to accelerate processes of the hydrological cycle in places like the humid tropics, observations that contribute to the understanding of resulting human-environment interactions and impacts are lacking (Wohl et al., 2012). Analyses of precipitation variability from gauge, satellite, reanalysis, and models of the Rio Negro basin have been conducted (Getirana et al., 2011, 2010; Juarez et al., 2009), and Getirana et al. (2011) conclude that gauge-based data are the most representative of actual precipitation. Since the northwestern Amazon is an area where there are low correlations for rainfall variations between datasets from the Climate Prediction Center (CPC), Global Precipitation Climatology Center (GPCC), Tropical Rainfall Measuring Mission (TRMM), and Global Precipitation Climatology Project (GPCP) (Getirana et al., 2011, 2010; Juarez et al., 2009), the potential of high spatial and temporal variability in this region may complicate climate change and sustainability analyses. Therefore, Indigenous participation in the research process is essential for evaluating impacts and distinguishing between socio-economic and climate forcings.

Given the importance of agroecosystems and fisheries production in the Tiquié basin, identifying timescales that relate to the slash-and-burn practice and fish reproduction and migration cycles is key to evaluating sustainability in this region. With the temporal framework of the Tukano ecological calendar documented by the AIMAs from 2005-2008, an observation-driven assessment is initiated. We combine indicators of ecological knowledge, precipitation, and river levels with cross-disciplinary methodologies for examining changes across different sites at multiple timescales.

4.1.2.1 Agroecosystems

Indigenous populations are finely attuned to the limitations of the Rio Negro black-water basin possessing profound knowledge of sustainable land-use practices that have been employed for centuries. Moran (1991) states that some of the most elaborate and effective adaptation strategies to environmental limitations can be found in the Rio Negro basin. For indigenous populations, these strategies include spatial mobility and multilocal land use, small ($\sim$50 people) dispersed settlements, cultivation of bitter manioc, and protection of flooded forests for fisheries (Moran, 1991). Travel by river is essential for managing fisheries and to reach cultivated lands.

Compared to other parts of the Brazilian Amazon, the upper Rio Negro region exhibits differences in ecosystem makeup, with low potentials for intensification of agriculture and fisheries. Some of the limiting factors in this black water ecosystem include oligotrophy or poor nutrient level and hydrological stress caused by cycles of floods and short-term droughts that last less than one week (Moran, 1991). Indigenous populations have managed scarcity of resources through a multilocal land-use system and rights negotiated through large kinship networks (Eloy, 2008; Moran, 1991). Gardens and productive backyards are controlled by domestic units, secondary growth and fallows by patrilineal and patrilocal groups, and dense forest is seen as a common use area and collectively managed (Eloy & Lasmar, 2011). Although agricultural products are often traded for fish and other items (like fish hooks, batteries, tobacco, etc.), almost all plots are intended primarily for sustaining families and not for commercial production.

The size of clearings and the length of cultivation are carefully managed through rotation taking into consideration nutrient levels and regrowth of native species, which can take 100 years in certain parts of this ecosystem (Moran, 1991). Historically, indigenous populations have avoided areas of heath forests on hydromorphic podzols, known as *caatinga do Rio Negro* or *campinarana*, and flooded forests, known as *igapós*, for agricultural use, giving preference to the so-called *terra firme* forests on oxisols and utisols.

Bitter manioc (*Manihot esculenta* Crantz) is one of the few crops that can be grown in the extremely acidic and nutrient-poor soils with toxic levels of aluminum. Although it is drought tolerant, going into dormancy and regaining its leaves when soil moisture returns, manioc must be cultivated on higher ground to avoid waterlogging (Moran, 1991; Fraser, 2010). This is the primary reason why flooded forests are avoided for agricultural use in the Rio Negro basin (van der Veld, 2014). The lifecycle of bitter manioc lasts between six months and two years, and certain varieties can be 'stored' underground for up to four years to provide for a continuous source of food (Fraser, 2010). As the staple ingredient for bread (*beijú*) and beer (*caixiri*), and used as a thickening agent in breakfast porridges and fish stews, it provides the bulk of the calories for indigenous populations of the upper Rio Negro region.

Women farmers maintain stocks of manioc varieties through mobility and exchange networks. They manage up to three plots at a time in different stages of maturation, with the average number of varieties planted per farmer ranging between 7 and 33 (Emperaire & Peroni, 2007). Newlywed women expand their stocks by joining varieties with their mothers-in-law, and varieties have been found to circulate several hundred kilometers away across the borders of Colombia and Venezuela. The key cultural importance of bitter manioc for the livelihood of the indigenous population of the upper Rio Negro is further evident in the 351 names collected by Emperaire & Peroni (2007). The majority of these names reflect the biological diversity of the area, including names of fish and other food crops, which are included in indigenous accounts of the origin of mankind and agriculture (Emperaire & Peroni, 2007). A high level of biodiversity of bitter manioc is, therefore, both a biophysical and a socio-economic indicator of sustainable agroecosystems in the upper Rio Negro. Since dry periods are necessary for drying out and burning agricultural plots, one of the few threats to the cultivation of bitter manioc is increased precipitation events which may alter possibilities for the slash-and-burn practice and, hence, the sustainability of these agroecosystems.

In recent years, there have been some reports that these dry periods have became more irregular, making it difficult for local families to plan their agricultural activities.

In the Tiquié basin, areas for cultivation usually range between an average of 0.3 ha and a maximum of about 1 ha. van der Veld (2014) measured the plots of six families in the community of São José and found that the average number of plots per family was 2.5 with the average combined area of cultivation on all plots totaling 0.7 ha. Plots cleared from primary forests are typically larger (Cardoso, 2010), though rare along the Tiquié (van der Veld, 2014). While smaller plots of forest regrowth, known as *capoeira*, that are closer to dwellings are rotated to fulfill immediate family needs, larger primary or secondary mature forest plots are farther away and require the help of younger men to clear.

According to the ecological calendar of the Tiquié, the times for clearing primary or mature secondary forest may occur anywhere between September and December, with clearing occurring usually two to several months before burning. Clearing of *capoeira* or forest regrowth can occur year round. In the annual cycle, burning may begin at the end of August with three to four days of direct sun necessary to dry out and burn a *capoeira* plot and one week or more required to dry out and burn a primary or mature-forest plot. Strong dry periods lasting up to one to two weeks, like *Mere Kuma* in December or January and *Use Kuma* in January or February, are usually necessary to burn down primary or mature secondary forest. Peaks in burning occur from December to March, while a lot less burning is possible in the wetter periods of May to August.

Although the slash-and-burn agriculture of indigenous peoples has been criticized as a contributing factor to greenhouse gas emissions (Palm et al., 2003), we ask the reader to consider the small scale at which this occurs compared to large-scale monocultures in other parts of the Amazon. Indigenous slash-and-burn agriculture creates a landscape of secondary forests in various stages of growth, mixed with primary forest. Primary forests have more biomass than secondary forests, and thus are greater carbon sinks. However, for every small parcel of forest indigenous peoples clear and burn, there is a much larger area of secondary forest growing and capturing carbon in the process.

4.1.2.2 Fisheries

The population of the Tiquié basin live in riverside fishing communities, and fish is an essential item of the daily diet. Indigenous peoples of the Rio Negro basin practice many forms of fishing - some occur only at certain times during the annual cycle and are specific to certain locations (like waterfalls and narrow, flooded channels of the rivers). Some fish traps are made so that fish come in, but cannot leave. The placement of traps is guided by a detailed knowledge of riverine geography and the ecology of fish species. The traps can be placed against or with the flow of water, and they can be fixed or placed and removed in accordance with river conditions.

Black water rivers are not very biologically productive, because they originate in poor soils of the heath forest. Black waters have very low mineral content and, consequently, reduced primary production (Goulding et al., 1988) compared to white water rivers. Flooded forest areas, such as *igapós*, are a river's main source of productivity through the mechanism of flood pulses (Bayley, 1995). These are important feeding areas for fish (Junk et al., 1997), where fruits, flowers, insects, etc. are readily found at the water surface. Several authors (Goulding et al., 1988; Chernela, 1994) have highlighted the interrelationship between the fish life cycle, fruit trees from riverbanks and *igapós*, and the regime of ebbs and flows. The rise of the river and the full flooding of the *igapós* facilitates the access of these resources by fish. Some types of traps are placed in channels or *igapós* to catch the fish during ebbs and flows.

The lower course of the Tiquié River is characterized by floodplains, *igapós*, and oxbow lakes. In the middle Tiquié, these features are intertwined with higher riverbanks and upland forests in areas of *terra firme*. In the upper course of the river, abrupt steep banks and areas of rapids and waterfalls predominate. The relief represented by waterfalls and rapids - in places like Pari–Cachoeira - serve as a barrier to the dispersal of many fish species.

The biological cycle of fish is closely related to fluctuations in river level, many phenological cycles, and other processes of the ecologial calendar. In the Tiquié, different intensities of fish migrations and gatherings for reproduction occur throughout the entire river. When the flood season is well-defined, the river grows and remains high for a prolonged period from April on, and the fish migrate in greater numbers. Otherwise, migrations are rather weak and the abundance of fish in the other months of the year is also lower. Reproduction occurs between November and April, especially during the months of January and March.

4.1.2.3 Climate Change in Northwestern Amazonia

The level of the Tiquié River is extremely variable, and closely follows the intense rainfall characteristics of the region. Throughout the Amazon basin, large-scale circulations and oscillations acting over a period of months to decades as well as patterns of mesoscale convective systems acting over a period of hours to days can influence interannual variability and result in extreme events in different parts of the basin. Manifesting at local to regional scales, some of the known influences on the climate of the Amazon and Rio Negro basins in recent years are here discussed.

The climate of the Amazon basin is influenced by large scale circulations and oscillations, namely the El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO), and sea surface temperatures (SST) in the tropical Atlantic which can influence the displacement of the intertropical convergence zone (ITCZ). A 3-4 year peak in Amazon basin rainfall and Amazon River discharge has been attributed to ENSO cycles, while a 24-28 year oscillation is attributed to the PDO with more precipitation seen in the Amazon basin between 1945-76 and less between 1977-2000 (Marengo, 2007). The active phase of the Madden-Julian Oscillation can also result in intense rainfall events in northern Amazonia (Waliser et al., 2012).

During the last decade, different parts of the Amazon basin have suffered extreme drought and flood events (Potter et al., 2011; Marengo et al., 2011, 2008). The droughts of 2005 and 2010 are two such events that have been well documented (Potter et al., 2011; Marengo et al., 2008). The drought in 2005 was attributed to the combination of an anomalously warm North Atlantic, a decrease in the moisture transport of the northeast trade winds in the summer season, and a corresponding decrease in convection and rainfall in that region (Marengo et al., 2008).

An overall intensification of the hydrological cycle in Amazonia since 1990 may be associated with increased SST in the tropical Atlantic (Gloor et al., 2013). In northwestern Amazonia, this increase is primarily occurring in the rainy season. In total, an excess of 80 mm per month was seen comparing the decades of 1981-1990 to 2000-2009 (Gloor et al., 2013).

Northwestern Amazonia has been identified as the highest precipitation center throughout the Amazon Basin with 3600 mm/yr (Marengo, 2007). Within this area, the Rio Negro basin exhibits interannual and subseasonal variability and the potential for rapid changes and high magnitude events. In both 2009 (Marengo et al., 2011) and 2012 (Satyamurty et al., 2013), record flood levels were recorded in Manaus port, where the Rio Negro meets the Amazon River. Satyamurty et al. (2013) found that interannual rainfall variability in the Rio Negro basin has increased since 1970 with an average positive trend of 10 cm/yr. The Rio Negro has the highest rate of runoff of all the Amazon tributaries corresponding to 4.36 mm/day (Marengo, 2004).

In a comparison between observations of river discharge and simulations from an enhanced version of the Dynamic Global Vegetation and Hydrology Model LPJmL for 1961-1990 and projections for 2070-2099, flooding duration in parts of northwestern Amazonia was found to increase by three months and shift by three months. The probability of three consecutive years with high precipitation, based on applying model results from 24 GCMs from the IPCC-AR4, also increased by 25% in this study (Langerwisch et al., 2013). These potential shifts in precipitation frequency and magnitude could have grave implications not only for the indigenous populations of northwestern Amazonia, but also for areas downriver in central and eastern Amazonia. In addition, areas in southern Brazil are directly affected by patterns of atmospheric moisture transport from Amazonia (Marengo, 2007).

4.2 Methods

Considering the complexity in the co-production of knowledge and the delivery of climate services to fit users' needs, a conceptual framework is conceived for the bridging of IK and WK to identify relevant spatial and temporal boundaries to reduce climate change impacts and vulnerability. This framework considers the importance of coexistence of IK and WK to evaluate and monitor environmental change. It employs novel, cross-disciplinary methodologies to combine qualitative and quantitative data and identify non-linear changes.

4.2.1 Participatory Research

In 2005, a series of meetings and workshops on environmental management were conducted by Tiquié River associations, involving significant participation of Indigenous leaders and community residents (from Brazil and Colombia) with support and advice from ISA, which has maintained a permanent team of advisors and researchers in the area since 1998. Cross-community agreements for environmental management were discussed for strengthening Indigenous governance of their territories. In these workshops, it was clear the multiplicity of factors associated with the dynamics of life cycles and the history of relations with the environment, both biological and socio-cosmological. One of the strategies of action was to start a team of researchers and community workers, selected by their communities to maintain, monitor and stimulate activities related to environmental management. These researchers and workers became known as indigenous agents of environmental management (AIMAs). The objective of the participatory research between AIMAs and ISA researchers was to promote IK, allied with WK, for understanding environmental and climate monitoring of the region by recording astronomical, ecological and socio-economic observations and improving environmental governance of this important biome (Cabalzar, 2010).

An AIMA is one form of community worker, working for best management practices, the collection of information and research. They are residents of the communities, of different age groups and levels of primary and secondary education. The number of AIMAs along the Tiquié River is between twenty to thirty. The AIMAs, like all residents of the communities, are involved in daily work and dedicated in part to research activities. As of 2014, there are five houses with solar energy, computers and meeting space to support research in different communities. The AIMAs participate in training programs and the exchange of knowledge through workshops coordinated with ISA. They receive a stipend proportional to their participation and training, plus some working tools and fuel required for travel.

The AIMAs were given notebooks, pens and drawing materials and a list of suggestions for observations and recordings, such as: (1) rainfall, (2) river level or ebbs and flows for navigation conditions, (3) season name in the Indigenous language, (4) phenology of important plants —cultivated and wild fruits and flowers —that are ripe and being consumed by people, fish and animals, (5) fish cycles —migration, diet, reproduction, and fisheries management, (6) reproduction and behavior of mammals and birds, and (7) reproduction of insects and amphibians (eg. flight of ants, spawning frogs, appearance of edible caterpillars, etc.). AIMAs were also advised to write daily, but each had complete freedom to do as they could and write in the language of their choice (Portuguese, Tukano, Tuyuka, Hupda, Yuhupda). Some who were former students of Catholic missions had learned to write only in Portuguese, while the younger AIMAs attended new Indigenous schools and had learned how to write in their language with recently developed alphabets.

Observations were recorded in communities along the Tiquié, which means that several AIMAs could have been recording simultaneously within the same or neighboring communities. This overlap in data recording limits possibilities of gaps that might occur with AIMAs who live in multiple communities or travel regularly throughout the region.

The concern to collectively systematize the information gathered daily led to quarterly workshops starting in 2006, bringing together the AIMAs with ISA researchers. In these workshops, lasting one to two weeks, daily observations are shared and a timeline is made for each region of the river. AIMAs carry out organization and analysis of data under the supervision of Elders or IK holders and ISA researchers. Elders are invited to discuss relevant issues from various points of view, and these discussions generate information for management and future research. At the end of the workshop, notebooks are collected to be scanned, typed, edited, systematized and summarized for the description and analysis of the annual cycle.

In this study, annual cycles from November 2005 to October 2008 (Figure 4.3) were compiled from fifty notebooks comprising more than 400 edited and typed pages. Most of these records were kept by Tukano and Desana AIMAs in the middle stretch of the Tiquié between Pirarara Poço and Pari–Cachoeira (Figure 4.2). If there was discrepancy regarding the identification of constellations

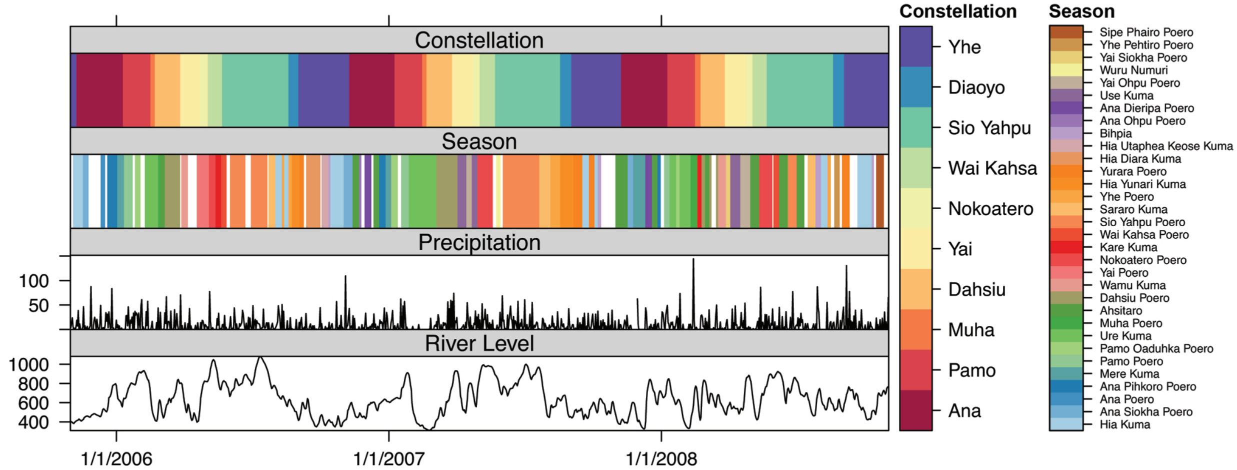

Figure 4.3: Example of the ecological calendar of the Tiquié based on constellations in Table 1 and seasons observed and recorded by the AIMAs from 2005-2008. Precipitation (mm) and river level (cm) data are from Agência Nacional de Águas (ANA). White areas in the season panel represent times with no observations.

Table 4.1: Primary constellations in the Tukano calendar and the date they start to set on the horizon at 20:30h (reference point is São Pedro, 0.27°N, 69.97°W).

Constellations										
	Aña (jararaca snake)	*Pamo* (armadillo)	*Muhã* (jacunda fish)	*Dahsiu* (shrimp)	*Yai* (jaguar)	*Ñokoatero* (Pleiades)	*Wai Kahsa* (fish rack)	*Sio Yahpu* (adze)	*Diaoyo* (otter)	*Yhe* (heron)
	Nov 8	Jan 9	Feb 15	Feb 21	Mar 27	Apr 23	May 3	May 22	Aug 19	Sep 2

or seasons in the notations of the AIMAs, in each case, the predominant constellation or season was decided based on a query to other materials, such as illustrations and notes of discussions between AIMAs and Elders at workshops.

4.2.2 Rain and River Gauge Stations

To begin evaluating climate change in the region, we utilized precipitation and river level data in conjunction with the Tukano ecological calendar. Precipitation and river gauge stations in the Rio Negro basin are installed and monitored by the National Geological Survey and the National Water Agency - Agência Nacional de Águas (ANA). These stations have been increasing since 1980, and observations are recorded by people hired in the local communities.

In general, gauge-based data are subject to observation errors, including systematic bias and random error, and data reporting from remote stations can be sporadic. However, since Getirana et al. (2011) concluded that gauge-based data are the most representative of actual precipitation for this area, precipitation datasets from ANA stations at Pari–Cachoeira (0.252°N, 69.784°W, 1980-present), Pirarara Poço (0.14°N, 69.21°W, 1992-present), and Taracuá (0.131°N, 68.541°W, 1961-present) were used for this study. ANA river level data at Pari–Cachoeira (0.25°N, 69.785°W, 1980-present), Cunuri (0.21°N, 69.38°W, 1982-present), and Taracuá (0.13°N, 68.54°W, 1977-present) were also examined. Although Taracuá is not on the Tiquié River, it was included in the study to offer an example of precipitation at the mouth of the Tiquié and to give a river level comparison from the larger basin of the Uaupés.

All precipitation and river level data, except for river levels prior to certain dates, are Level 1 data processed and made available on the ANA website (http://www.ana.gov.br/PortalSuporte/frm SelecaoEstacao.aspx). River levels prior to August 2006 for Pari–Cachoeira, September 2006 for Cunuri, and January 2008 for Taracuá are Level 2, which constitute organized raw data. Time series of precipitation starting in November of 1962 for Taracuá, 1982 for Pari–Cachoeira, and 1992 for Pirarara Poço and all continuous through October 2012 are plotted in Figure 4.4 along with river level time series from November 1982 through October 2012. Missing values for precipitation in Pari–Cachoeira (10.50%), Pirarara Poço (0.90%), and Taracuá (1.86%) were filled with zeros in order to have a continuous time series for the wavelet multiresolution analysis, while missing values for river levels at Pari–Cachoeira (6.39%), Cunuri (0.54%), and Taracuá (0%) were filled by linear interpolation. The only missing data during the 2005-2008 period occurs for precipitation at Pirarara Poço from November 23 to 27, 2007.

4.2.3 Classification Trees

In an effort to join the results of the Tukano ecological calendar with the rainfall and river level data, we used a classification tree approach. Classification trees use numerical techniques for exploring data and provide a graphical representation that can be used for description and prediction of complex processes or patterns. Given combinations of explanatory variables (i.e. constellation, river level, precipitation, dry days), classification trees repeatedly split data into more homogenous

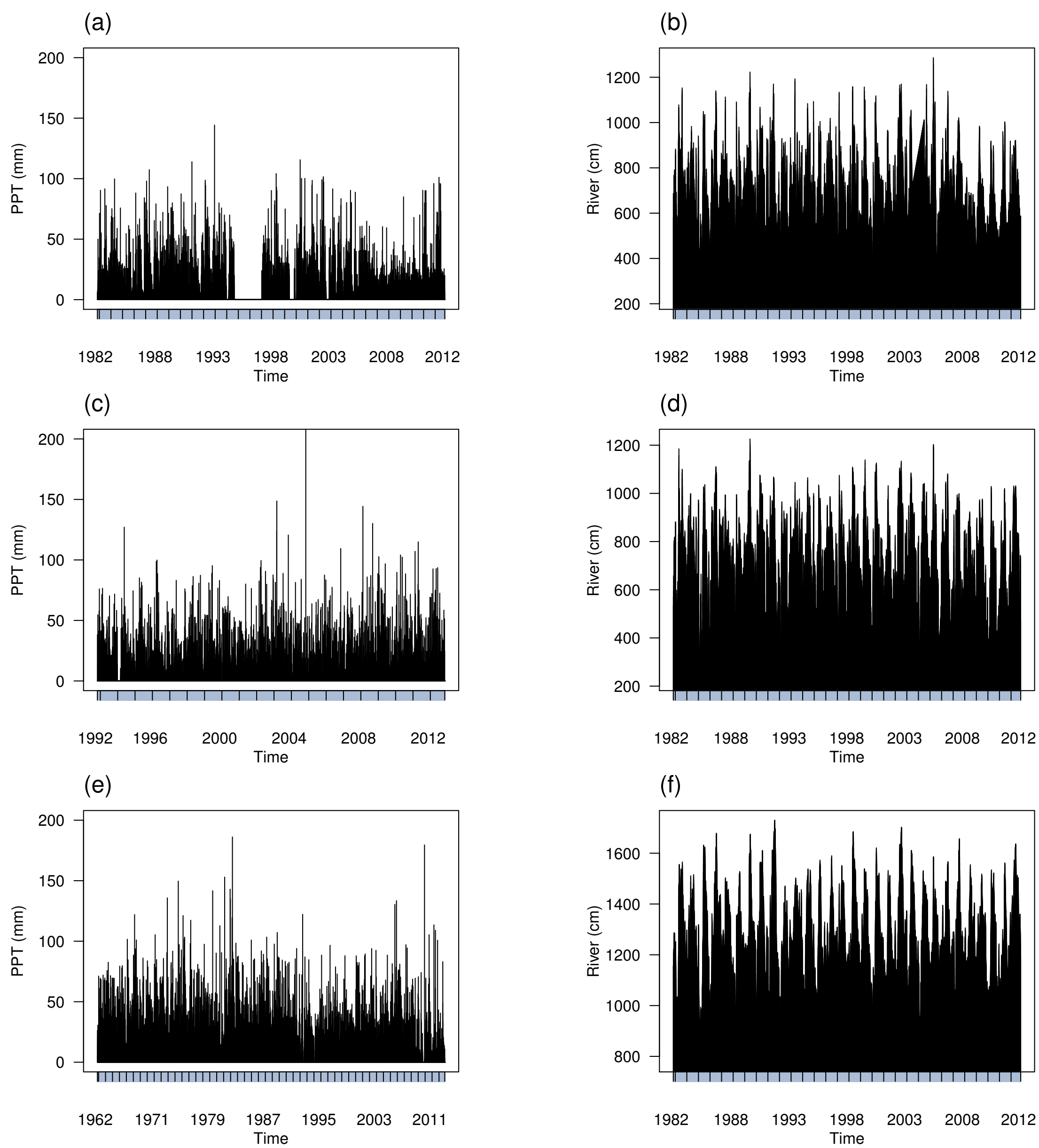

Figure 4.4: Time series of precipitation at (**a**) Pari–Cachoeira, (**c**) Pirarara Poço and (**e**) Taracuá, and river levels at (**b**) Pari–Cachoeira, (**d**) Cunuri and (**f**) Taracuá.

groups to get at the response variable (season). In other words, a group can be characterized by the distribution of the response variable, total observations in a group, and the values of the explanatory variables. Within a classification tree, splitting criteria can be defined with common indices, like the information entropy index and the Gini index, taking into account the proportion of the absences and presences in a group (De'ath & Fabricius, 2000).

To explore defining variables of seasons witnessed during 2005-2008, classification models were created using the rpart library in the R statistical software (Therneau et al., 2010). Categorical data of seasons and constellations where combined with numerical data for precipitation, river level, and consecutive days without precipitation at Pirarara Poço. The initial formula was Seasons=0.64Constellations (astro)+0.31River Levels (river)+0.01Precipitation (ppt)+0.04Dry Days (dry_days), but was then simplified to Seasons=0.18Precipitation+0.82Dry Days to get at a classification that gave importance to days without precipitation. Dry days were defined through quartiles for all 2005-2008 precipitation data, with everything below the 50% quartile value of 1.8 mm/day considered no precipitation. This threshold was selected because 42% of days had 0 mm of precipitation, while 4% had less than 1 mm and 3% had between 1 and 1.8 mm. Splits in the classification tree were based on the Gini index, where the largest category is typically separated in a split.

4.2.4 Wavelet Multiresolution Analysis

Once we examined defining variables for seasons in the Tukano calendar during 2005-2008, a longer-term analysis was applied to identify relevant timescales of precipitation and river level variance. Wavelet multiresolution analysis (WMRA) is a powerful tool that can identify changes occurring over time based on contributions from each frequency (or period) of a time series. It is a technique that can be applied across disciplines and has gained popularity in applications for hydrology and the geosciences (Kumar & Foufoula-Georgiou, 1997; Ruddell & Kumar, 2009; Stoy et al., 2009; Brunsell, 2010; Cochran & Brunsell, 2012). The reader is referred to Torrence & Compo (1998) for original equations, but a brief outline is provided here.

The basic idea of the analysis technique is to break down a time series (e.g. precipitation or river level) into a series of different timescales such as daily, weekly, seasonal, annual, etc. This decomposition allows us to analyze various components of the data that are not easily ascertained from the original data set. For example, if the annual scale is becoming more important, this would be reflected through an increase in the power (or variance) at the annual timescale.

In this study, wavelet renderings of precipitation and river level time series were examined at each scale of decomposition for temporal variations. This decomposition of the original data is conducted through the dilation (expansion and contraction) and translation (shifting along in time) of a "mother" wavelet, such as the Morlet wavelet used in this study. The wavelet analysis is ideal for the analysis of non-stationary signals because it exhibits a "zoom-in" capability that allows for the identification of brief, high-frequency events and low-frequency variability simultaneously (Lau & Weng, 1995).

For the time series of precipitation and river level, the continuous wavelet transform (wt) function is applied from the biwavelet library in R (Gouhier & Grinsted, 2014). The normalized, bias-corrected power is computed and plotted using the method described by Liu et al. (2007). The continuous wavelet transform results in a wavelet power at each point in time for each scale considered in the analysis (e.g. Figure 4.7). These colorful plots illustrate a full range of time series information that can be examined either across temporal scale at a single point in time, or across time at a single scale. The dotted white lines denote the cone of influence (COI) outside of which results are unreliable due to edge effects near the beginning and end of the time series. The black lines outline the areas of 95% significance obtained from a regular χ^2 test.

By examining the results at a specific point in time, we can tell the relative contribution of low frequency (i.e. annual) and high frequency (i.e. daily) cycles to the observed precipitation or river level at that time. By examining a single timescale through time (e.g. Figure 4.8), we are able to quantify how the overall contribution of that timescale is changing over time. The trends of the wavelet power time series can also be examined and plotted about the mean of the time series at selected scales (e.g. Figure 4.9).

The purpose of this analysis is to investigate the role of changes in different time scales. For example, if there is a regularly occurring cycle at the 4-year timescale (e.g. ENSO), we would expect to see periods of high power followed by a period of low power at the 4-year timescale in a fairly predictable cycle. If, however, there was an intensification of the hydrological cycle occurring at the 4-year timescale, we would expect to see a trend in the power as time progressed. Here, we quantified the trends at selected scales in order to quantify to what extent there has been an observable intensification in the precipitation and river level datasets. In addition, we utilized the scale information at each time to examine how the relative contribution has changed in selected periods.

In addition to examining the variability in a single time series, wavelet analysis also allows for quantifying the interaction between two time series. This provides useful information concerning which time series is leading the other and to what extent the time series are in phase with one another. Here, the wavelet coherence transform (*wtc*) was computed between precipitation and river level time series using the method described by Veleda et al. (2012) to calculate the bias-corrected cross-wavelet power. On the *wtc* plots, arrows pointing down signify that precipitation leads river level by $\pi/2$.

4.3 Results and Discussion

4.3.1 Tukano Ecological Calendar

For the Indigenous peoples of the northwestern Amazon, knowledge and management practices are linked over annual cycles. As they describe the annual cycles, IK holders of the Tiquié reference astronomical constellations, most of them at the celestial equator. Each constellation has a native narrative associated with an episode of the creation of the world. The constellation considered in each period is one that is setting in those days in the early evening when it is already visible at dusk (Cabalzar, 2010). In other words, the Tukano ecological calendar is a sidereal calendar with a cosmical setting.

There are ten principal constellations, shown in Table 1 and Figure 4.3, plus a variable number of other less significant ones for the identification of wet periods. Most of the constellations are named after animals, some are tools, and the Pleiades are sometimes represented as a group of stars or a figure. The larger constellations, like the *jararaca* snake and the jaguar, are divided into several parts to improve time measurement.

The calendar emphasizes the hydrological cycle (precipitation and above all the fluctuations in the level of the river and its tributaries), fish life cycles (especially species from genus Leporinus), and agricultural activities. The cycle of constellations is combined with events, like the rising of the rivers and the flooding of the surrounding forest, called *Poero* in the Tukano language, and short dry periods that last between four days to approximately two or three weeks, known as *Kuma* in the Tukano language. Therefore, "seasons" are roughly differentiated into two categories: wet periods and dry periods (we use the term "seasons" as synonymous with "periods" throughout this paper, although neither word may have meaning beyond our Western context). Wet periods are characterized by heavy rain and days without ebb of the river, while dry periods are characterized by sunny days and ebb of the river. While wet periods are named according to the constellation setting in the early evening, dry periods are mostly named according to a phenological occurrence, such as the ripening of cultivated fruits like peach palm, *cucura*, *umari*, *abiu*, and *ingá*. As an example, the wet period *Yai Poero* is named after the jaguar constellation, *Yai*, while the dry period *Mere Kuma* is named after the ingá fruit.

The Tukano year begins with the fall of the constellation of the *jararaca* snake, *Aña*. This astronomical phenomenon happens in the second week of November in the Gregorian calendar, which is why in this study the annual cycle starts in November.

While the constellations can serve as a fixed temporal framework, the seasons that accompany them are based on phenomena and biological cycles that can vary a few days to a few weeks, from year to year. Therefore, the seasons are not predictable throughout the annual cycle, and some seasons may be absent in a given year. This is evident in the annual cycles documented for 2005-2008 shown in Figure 4.3.

4.3.2 2005-2008 Data and Classification Trees

In this study, one way in which the Tukano calendar and climate data of precipitation and river levels are joined is through the use of classification trees. Based on data of constellations, river levels, precipitation, and consecutive dry days that occurred during the sample period of 2005-2008, the classification tree shown in Figure 4.5a attempts to determine seasons recorded by the AIMAs. Of the 32 seasons represented across all three years, only 11 of the most prominent across the three-year sample were defined in the classification tree for Pirarara Poço (Figure 4.5a).

Referring back to Figure 4.3, we can see the distribution of the 32 seasons in 2005-2008 and annual variations. The classification trees help us to summarize this data by pulling out the most prominent seasons and their defining constellations and river level thresholds. For example, prominent wet periods for Pirarara Poço include: *Dahsiu Poero* which occurs under the *Dahsiu* constellation when the river is greater than or equal to 416 cm, *Sio Yahpu Poero* which occurs under the constellation of *Sio Yahpu* with river levels higher than 686 cm, *Yurara Poero* which occurs under the *Yhe* constellation with river levels greater than 480 cm, and *Ñokoatero Poero* which occurs under the constellation of *Wai Kahsa*. One of the strongest dry periods at Pirarara Poço, with as many as 5 (2006), 15 (2007), and 7 (2008) consecutive dry days during our sample period is *Ure Kuma*, occurring under the constellations of *Pamo*, *Muhã* and *Dahsiu*. In 2008, *Ure Kuma* occurred only under the constellation of *Pamo*. *Mere Kuma* with up to 9 consecutive dry days occurs under the constellation of *Aña* when the river level is less than 651 cm. Two short dry periods with up to 4 or 5 consecutive dry days are *Use Kuma* occurring under the constellations of *Ñokoatero* and *Yai*, and *Hia Kuma* under the constellations of *Aña*, *Sio Yahpu*, and *Yhe* with river levels below 480 cm and above or equal to 525 cm in our sample period.

Given our three years of data for seasons, precipitation amounts in Pirarara Poço and consecutive dry days, our classification model clearly identifies 3.5 as the number of consecutive days without precipitation that differentiates a dry season like *Ure Kuma* from a wet season like *Sio Yahpu Poero*. This corresponds with knowledge of Indigenous agroecosystems where a period of 3-4 days is needed to dry out small capoeira plots for burning. The percentages in Figure 4.5b

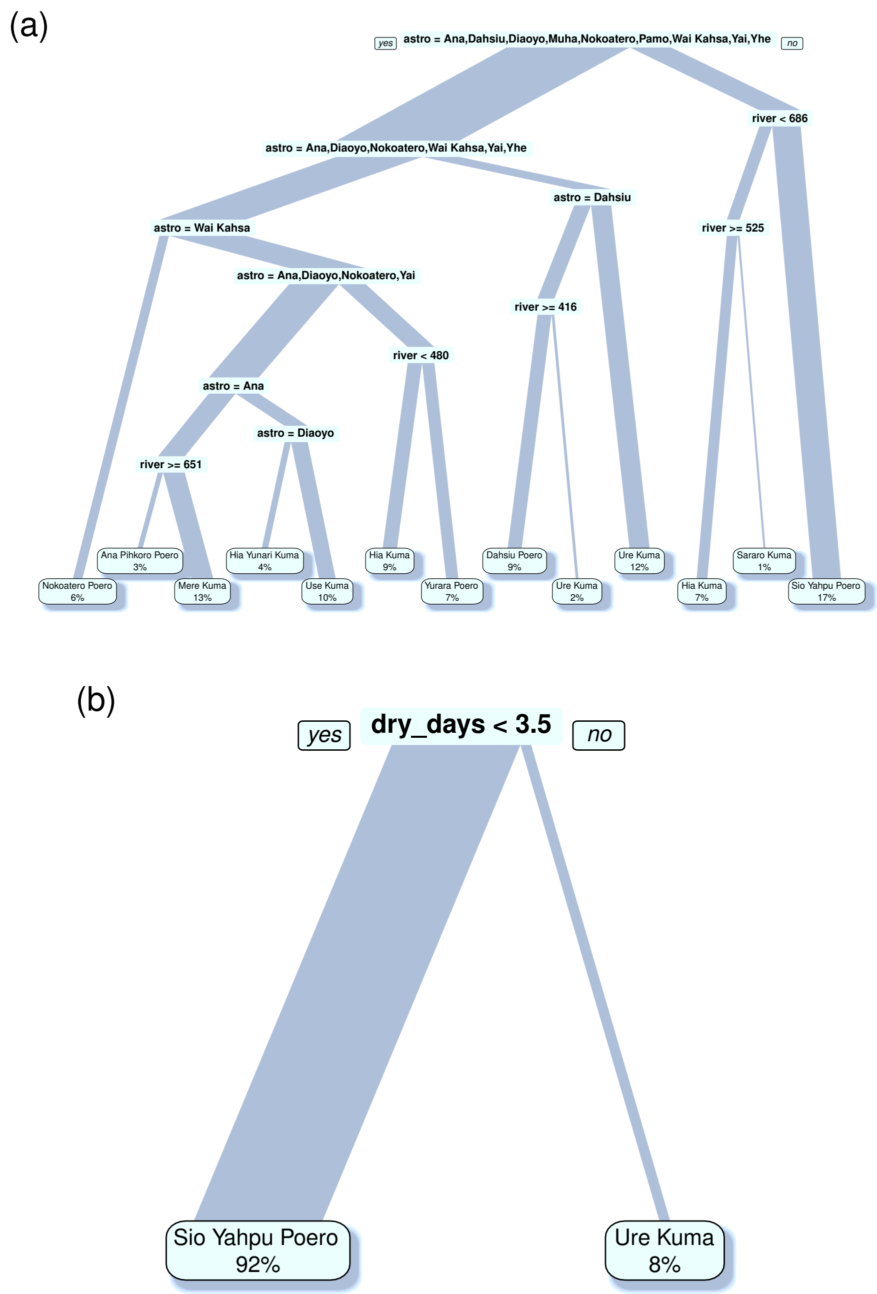

Figure 4.5: Classification trees for (**a**) Seasons=0.64Constellations (astro)+0.31River Levels (river)+0.01Precipitation (ppt)+0.04Dry Days (dry_days), and (**b**) Seasons=0.18 Precipitation (ppt)+0.82Dry Days (dry_days) at Pirarara Poço using data corresponding to seasons illustrated in Figure 4.3.

(92% for *Sio Yahpu Poero* and 8% for *Ure Kuma*) give us an indication of how infrequent a period of 3.5 consecutive days with no precipitation is during a three-year period.

Overall, the role of constellations in defining wet seasons known from participatory research is evident in the classification tree for the three-year sample. However, the classification tree does not provide for a clear classification of all wet and dry seasons. This may be due to the inherent variation in seasons from year to year as well as the lack of incorporation of phenological data, which as the participatory research has identified are essential for distinguishing dry seasons. Therefore, the fact that we have used these criteria: season name, constellation name, river level, precipitation, and consecutive dry days, does not necessarily mean they are the ones that define constancy in the Tukano ecological calendar. A limitation of this analysis is that we still have not included data for phenological cycles and other ecological or socio-cultural phenomena that might also influence the sequence of seasons. Nevertheless, the classification trees suggest that river levels play a role in distinguishing between wet and dry seasons, and that consecutive dry days only seem to influence the distinction between a wet and dry season when constellations or river levels are not considered. Although this may not reflect reality, it highlights the complexity of the Tukano ecological calendar and begs the question of how representative this three-year sample period is of the interannual variability for the region.

4.3.3 Examining Scales with Wavelet Analysis

Long-term time series of precipitation and river levels allow us to examine the spatial and temporal differences as well as the interannual variability in these records (Figure 4.4). Wavelet analysis was used to quantify the temporal changes at each site, thus facilitating an analysis of the individual timescales across sites.

The average wavelet spectra for precipitation shown in Figure 4.6a give us a way of looking at the wavelet power averaged over the entire time series from each site and plotted against scales from 2 to 4096 days. This facilitates an understanding of the overall contribution of each timescale to the total dataset. Here, we can see that the precipitation is dominated by shorter timescales (with

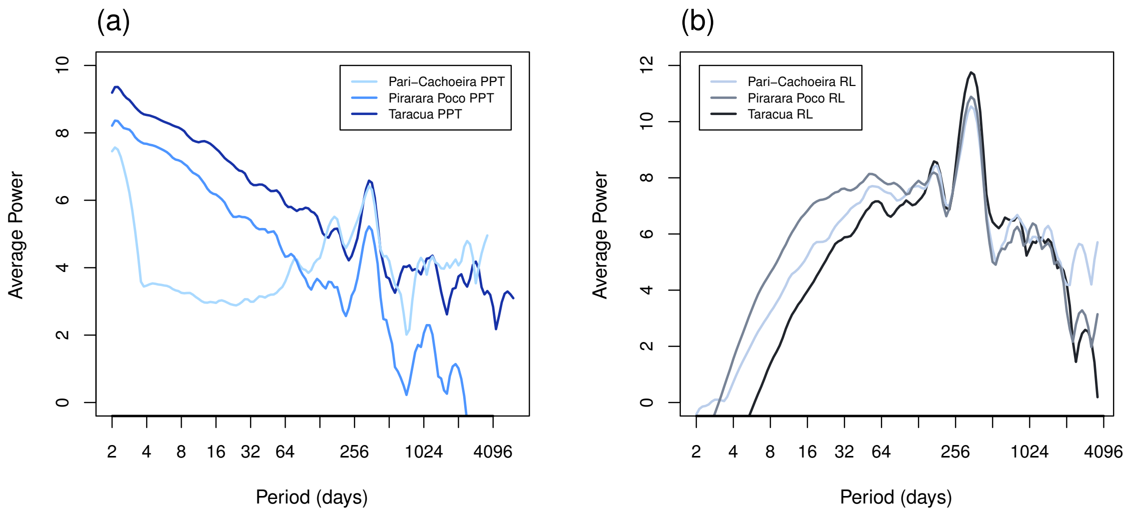

Figure 4.6: Average wavelet power (corrected and normalized) for all timescales for (**a**) precipitation time series and (**b**) river level time series.

increasing power up to the 2-day timescale) at Taracuá and Pirarara Poço, with more variance at Taracuá. We can also note what is presumably the impact of the gap filling of the missing data on the 4 to 64-day timescales in the Pari–Cachoeira site.

While Taracuá has the highest variance in precipitation, this translates to the lowest variance in all but the annual scale in the average power spectra for river levels in Figure 4.6b. This illustrates the idea of the basin as an effective integrator of precipitation, which highlights the importance of examining both precipitation and river level data when analyzing local and regional variability. All river levels exhibit the greatest power or variance at the annual scale. At the 1024 to 2048-day scale, we see some interesting divergences in both precipitation and river level suggesting that Pari–Cachoeira may be more influenced by a 3-5 year event than the other two sites.

Now that we understand the overall variability in the timescales across sites, we can analyze the scalograms to assess how these relative contributions vary across time. In Figure 4.7, the wavelet scalograms of precipitation are shown on the left while the wavelet scalograms of river levels from corresponding sites are shown on the right. Significant wavelet powers are outlined in black (significance level of 95%) and anything below the COI (white dashed line) is not necessarily reliable and, therefore, not discussed. Areas of dark blue represent missing data filled according to the methods above and primarily occur in the Pari–Cachoeira time series.

Precipitation offers a local signal whereas river levels translate to a more regional, basin-wide signal (Figure 4.7). Precipitation time series exhibit significant variance at the 2-day to weekly scales, with some significant times in the 8-day to 64-day periods, and swaths of significant variance across most of the time series at the annual scale (Figure 4.7a, c, and e). One exception is the lack of significant variance at all scales in precipitation from approximately 2005-2010 at Pari–Cachoeira.

The wavelet scalograms of river level show a difference in variance between the Tiquié and Uaupés Rivers, especially at shorter timescales (Figure 4.7 b, d, and f). Typical of a lag between precipitation events and river level change based on channel and basin size, significant variances are especially prominent from the 4-day to the 16-day scale along the Tiquié (Figure 4.7b and d) and the 8-day to 32-day scale along the Uaupés (Figure 4.7f). From the 32-day to the annual scale significant variances are more dispersed, especially for the Tiquié stations. The variance at the annual scale for Pari–Cachoeira and Cunuri show significance around 1986, early 1990s, 1997-2003, and 2010, implying that regional events (environmental and/or socio-economic) may have occurred that influenced both sites. From 2005-2010, Cunuri shows more significant variance than there is upriver at Pari–Cachoeira, suggesting a more local event at either location. During this time, Pari–Cachoeira also exhibits a drop in variance at shorter timescales, which is not apparent at Cunuri.

When we examine wavelet power time series for Pari–Cachoeira and Pirarara Poço scale by scale (Figure 4.8), we can more clearly see how variance in precipitation or river level at short timescales contribute to the annual scale. For example, an increase in precipitation variance at the annual scale in Pari–Cachoeira from 2010 to 2012 may be related more to an increase in 4-day power than 16-day power (Figure 4.8a). This is confirmed in Figure 4.9a, where we examine the wavelet power time series trend and how it differs from the mean (red dashed line). For 2010-2012, the 4-day trend is far above the mean compared to the 16-day trend, which implies that the anomaly in the 4-day trend is contributing most to the annual increase in variance in those years.

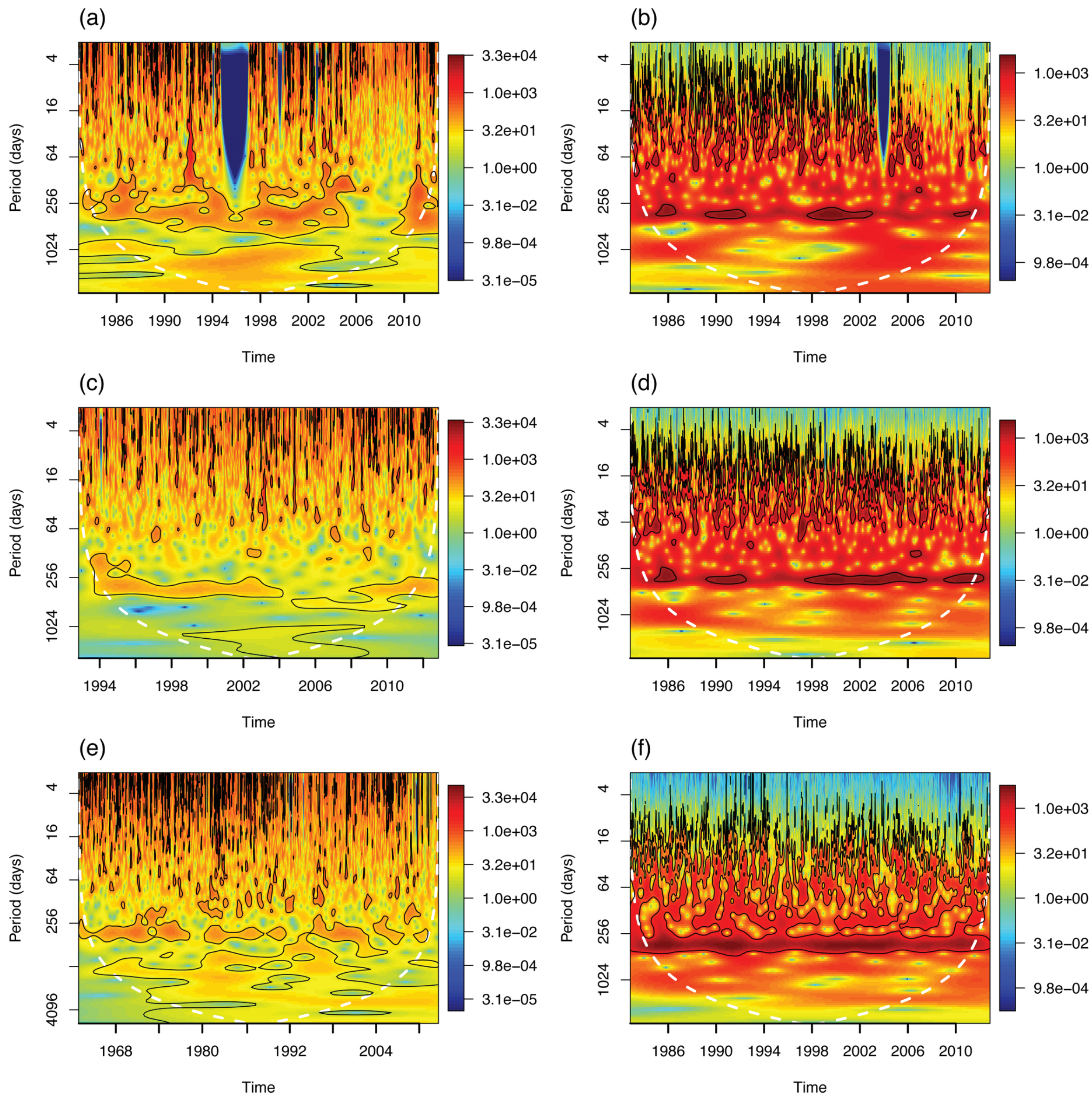

Figure 4.7: Wavelet transforms (*wt*) of precipitation at (**a**) Pari–Cachoeira, (**c**) Pirarara Poço and (**e**) Taracuá, and river levels at (**b**) Pari–Cachoeira, (**d**) Cunuri and (**f**) Taracuá.

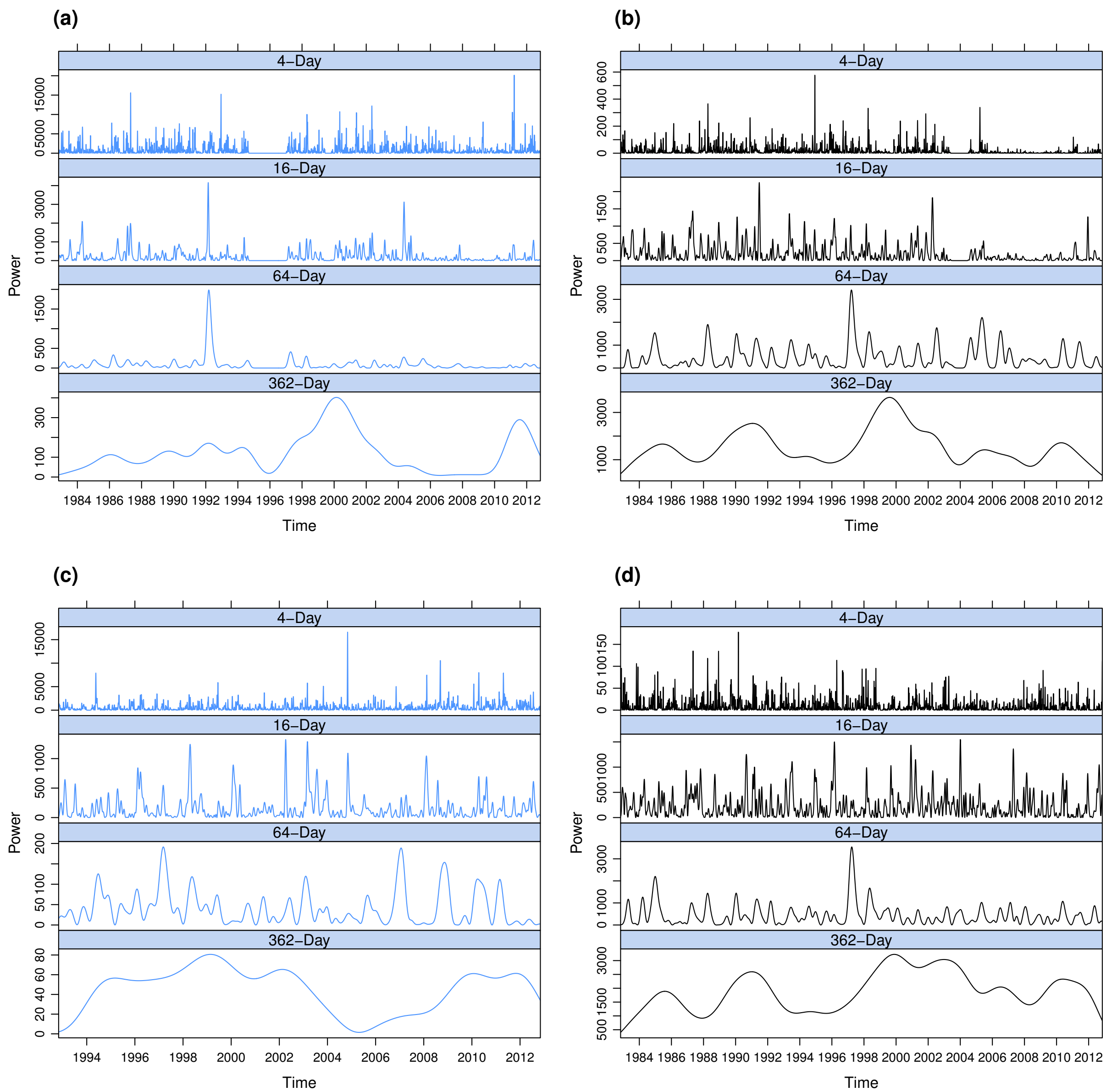

Figure 4.8: Wavelet power (corrected and normalized) for 4-day, 16-day, 64-day, and 362-day (annual) timescales for precipitation at (**a**) Pari–Cachoeira and (**c**) Pirarara Poço, and for river levels at (**b**) Pari–Cachoeira and (**d**) Cunuri.

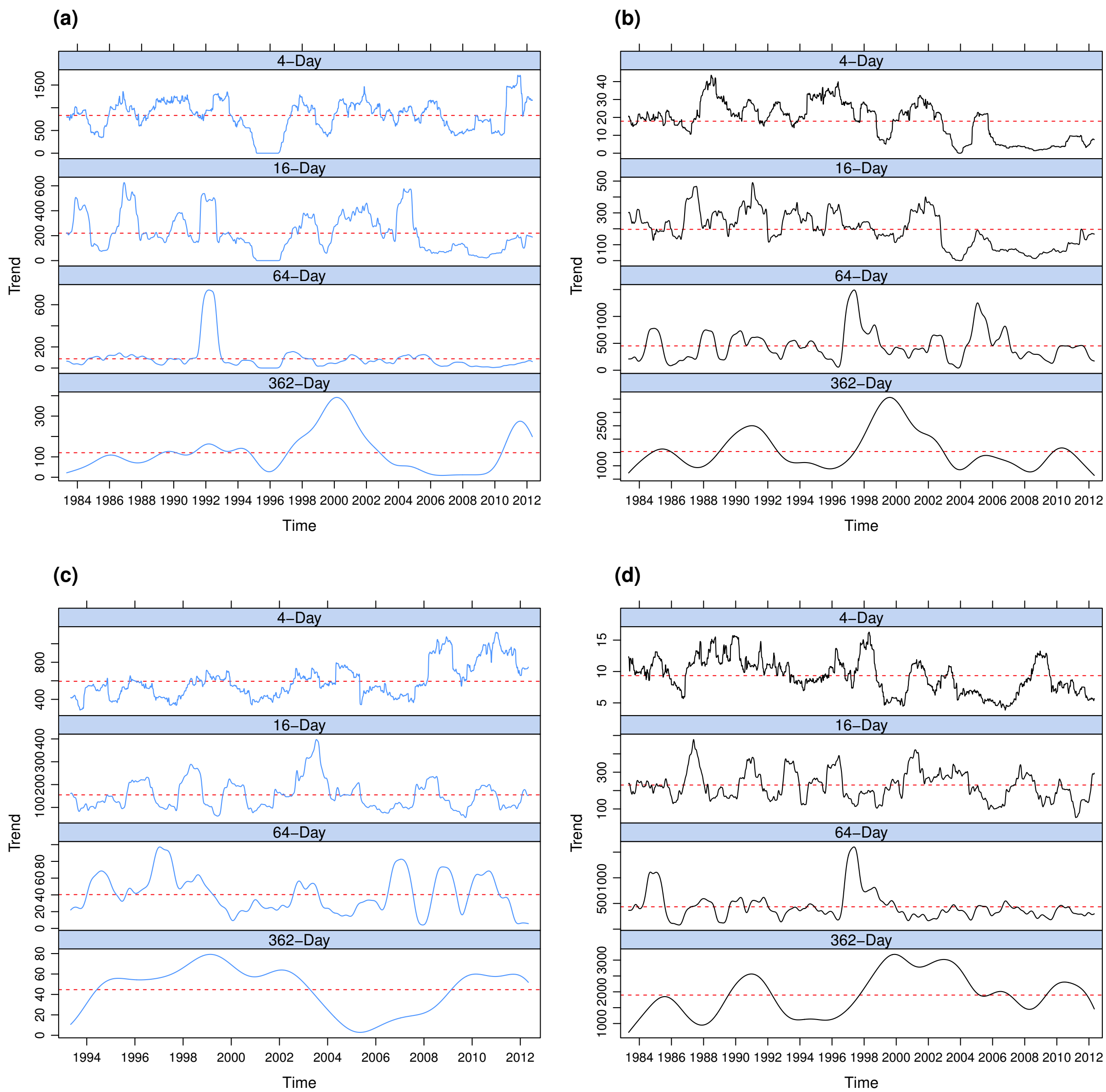

Figure 4.9: Trends from wavelet power (corrected and normalized) for 4-day, 16-day, 64-day, and 362-day (annual) timescales for precipitation at (**a**) Pari–Cachoeira and (**c**) Pirarara Poço, and for river levels at (**b**) Pari–Cachoeira and (**d**) Cunuri.

River levels at both Pari–Cachoeira and Cunuri tend to have higher trends prior to 2000 and lower trends after 2000 at shorter timescales. While this switch in Tiquié River level trends could be related to PDO (Marengo, 2007), it could also be indicative of regional influences related to sea surface temperatures in the Tropical Atlantic as discussed by Gloor et al. (2013). A low trend in river level variance below the mean at Pari–Cachoeira from 2006-2011 is seen at the 4 and 16-day scales (Figure 4.9b). This low variance indicates consistent high or consistent low river levels at these timescales. Given that the fluctuations in river levels throughout the year determine times of fish migration and reproduction, alterations in these patterns could impact the sustainability of fisheries and influence household food security.

The results from Figures 4.8 and 4.9 show how we can narrow down changes occurring to relevant timescales, in this case those related to wet and dry periods known through the Tukano ecological calendar. Although it is difficult to pinpoint in what season or under what constellation changes may be occurring based on our three-year sample, we know that the 4-day scale is important for distinguishing between wet and dry periods and for the agricultural practice of slash-and-burn. Therefore, if we look at the increase in variance of 4-day precipitation occurring at Pirarara Poço starting in 2008 and Pari–Cachoeira starting in 2010 (Figures 4.9c a), we can speculate that this increase could impact agroecosystems. An increase in variance of precipitation every 4 days could affect both capoeira and mature forest agricultural plots, whereas a change in 8-day or 16-day variance might only affect mature forest plots.

To examine the interactions between the precipitation and river levels at corresponding stations, we inspect the wavelet coherence transforms in Figure 4.10. Precipitation and river levels show strong correlations at the annual scale at all sites except for during the years of 2002-2006 at Pari–Cachoeira (partially due to missing data) and at Taracuá, and for 2004-2008 at Pirarara Poço. This period of time happens to coincide with the 2005-2008 records of the AIMAs for the Tukano ecological calendar, which indicates that the seasons described in those years may not be representative of the long-term trend. The low trend in river level at Pari–Cachoeira from 2006-2011 that accompanies the lower trend in precipitation at all scales during that same time is observable in

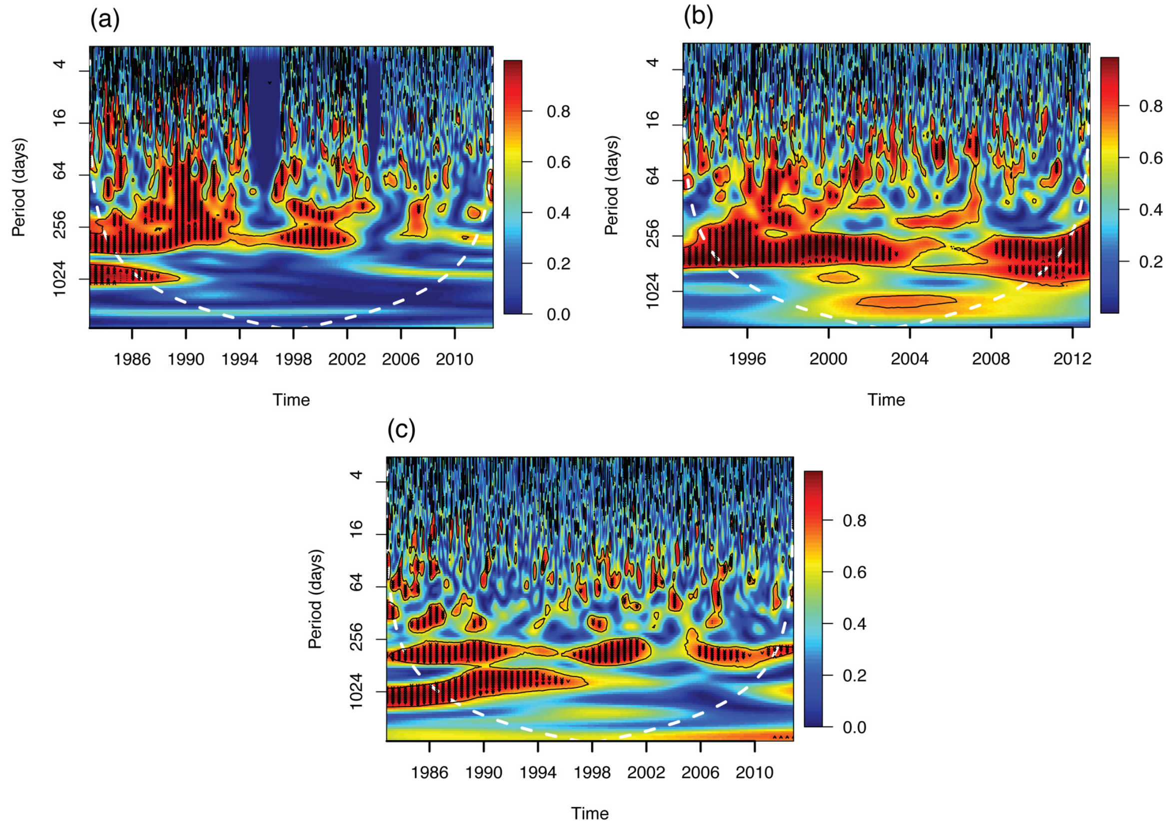

Figure 4.10: Wavelet coherence transform (*wtc*) between precipitation and river levels at (**a**) Pari–Cachoeira stations, (**b**) Pirarara Poço and Cunuri stations, and (**c**) Taracuá stations.

the wavelet coherence transforms (Figure 4.10a). Although the arrows pointing downwards indicate that precipitation leads river levels at all sites, it is important to keep in mind that the Cunuri river-gauge station is upriver from the precipitation measurements at Pirarara Poço.

Times when precipitation is decoupled from river levels at nearby stations may imply a socio-economic forcing, such as the creation of a dam, versus a climatic one, such as changes in the Tropical Atlantic SST, MJO, ENSO or PDO. In addition, although deforestation in the Tiquié is very minimal compared to other parts of the Amazon basin, Spracklen et al. (2012) have found that the passage of air over extensive areas of forest over a period of days can produce at least twice the amount of precipitation as air that passes over deforested areas. This is due in part to the role of forests in precipitation and evaporation recycling ratios, which van der Ent et al. (2010) estimate to be up to 28% of precipitation and 48% of evaporation annually for the Amazon region. Consequently, deforestation patterns throughout the Amazon could create changes in rain patterns over forested areas, such as the Tiquié basin.

4.4 Conclusions

Throughout the Tiquié basin, high spatial and temporal variability in precipitation and river levels are likely to complicate climate change and sustainability analyses. In an initial attempt to address this complexity, Western researchers have attempted to bridge Indigenous and Western knowledge systems to provide climate services for sustainable livelihoods and orient future research. Using methodology that could be employed with other Indigenous or non-Indigenous astronomical and ecological calendars, timescales of relevance for evaluating and monitoring sustainability in agroecosystems and fisheries have been identified.

With wavelet analysis, we have isolated a recent trend in the increase of precipitation variance at the 4-day scale at Pirarara Poço. This scale, also identified by the classification trees, is important to the slash-and-burn practice in Indigenous agroecosystems for bitter manioc production, since 3-4 days of sun are needed to dry out and burn small capoeira plots and 7-10 days are needed to dry out and burn larger plots of primary or secondary mature forest. Future research could explore climate change and socio-economic impacts to precipitation at this 4-day scale and why changes may be occurring at Pirarara Poço and, to a lesser extent, at Pari–Cachoeira.

Precipitation and river level trends in variance remained low at Pari–Cachoeira from 2005 to 2010. Since river levels give us a more regional indication of precipitation, there may be a greater possibility of a climatic forcing influencing this period of time. Wavelet coherence transforms can be used to further explore teleconnections with SST in the Tropical Atlantic, MJO, ENSO, and PDO.

Incorporation of phenological data may improve definitions of dry periods in the Tukano calendar, particularly since the naming of these seasons is so tightly coupled to phenological processes. The tremendous undertaking of sorting through and summarizing the AIMAs notes is a slow process, yet it results in a wealth of knowledge that may not be captured by instrumentation. On the other hand, instrumentation for georeferencing of data collected by AIMAs could facilitate future evaluations of spatial and temporal variability.

Our methodology and use of long-term data help us see that recorded observations of the Tukano ecological calendar for 2005-2008 occur during a time where there was relatively low variance in precipitation and river level at the annual scale, which is not characteristic of the long-term trend in the region. What is currently available in the 2005-2008 record of the Tukano calendar is indicative of its future utility for climate change and sustainability assessments in Indigenous territories. Essential to these assessments is the continued Indigenous participation in systematic data collection and interpretation of results for distinguishing between socio-economic and climate forcings and evaluating climate impacts.

Efforts to join Indigenous and Western knowledge systems are vital for sustainable environmental management in Indigenous territories and other regions where traditional management and systems of responsibilities may be challenged in the context of global climate change (Berkes, 2006; Whyte, 2013). Climate change is a not only a justice issue for Indigenous peoples, but also for ribeirinhos, caboclos and other traditional peoples in Brazil and around the world who depend on land or riverine systems for their livelihoods. This study contributes to theoretical and conceptual frameworks for bridging diverse knowledge systems by leveraging the work of boundary organizations (Guston, 2001) and addressing the coexistence of diverse knowledge systems and varying spatial and temporal scales together (Berkes, 2006).

While the coexistence of IK and WK systems in this study identified relevant timescales, it remains unclear whether this climate information will be useful in supporting collective continuance or Indigenous adaptive capacity and reducing climate change impacts and vulnerability. It is our hope that the methodology employed has resulted in knowledge that can be more easily incorporated into the Tukano cultural systems (Davis, 2006). Since the delivery of climate services must go beyond peer-reviewed publication (McNie, 2013), our findings will soon be translated to Portuguese and Tukano. Results will be disseminated in oral, written and visual formats and discussed at workshops to build adaptive capacity and support emerging systems of responsibilities in Tiquié communities. Yet, “integration” of WK with IK will only occur if AIMAs and their communities decide the information from the Western climate analysis is useful for planning agricultural and fisheries activities.

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Chapter 5

Conclusion

From drought in the US Great Plains, increased precipitation frequency in northwestern Brazil, and an increased urban heat island in a mega-city in southeastern Brazil, this dissertation research demonstrates some of the ways in which climate change and sustainability intersect. By exploring different biophysical and ecological indicators, and explicitly accounting for spatial and temporal scale, the three case studies quantify environmental sustainability and, not surprisingly, find an overall move away from biophysical sustainability. In so doing, they also reveal pathways that can be followed to move agricultural, urban or forest systems back toward greater sustainability. These pathways can contribute not only to increased sustainability but also to climate change mitigation and adaptation at local to regional scales resulting in win-win situations. Additionally, the findings of each study here summarized contribute to four of the 17 Sustainable Development Goals (SDGs), offer inputs for policy considerations, and build on theory and conceptual frameworks for future applications of sustainability science.

5.1 Summary of Findings

5.1.1 Chapter 2

Combined with indicators of net ecosystem exchange of carbon (NEE), soil water content (SWC), and the Bowen ratio, the entropy production (σ) metric in Chapter 2 provides a good indication of whether a system is moving away from or toward thermodynamic equilibrium and sustainability. Irrigated maize stressed by hydrologic and high temperature anomalies associated with the 2012 US drought exhibits the greatest increase in σ, indicating decreased sustainability compared to rainfed croplands and grasslands. These results suggest that maximizing carbon uptake with irrigation (or fertilizer) use tends to move agroecosystems away from thermodynamic equilibrium and sustainability, which has implications for food security and greenhouse gas (GHG) mitigation for climate-smart agriculture.

Though irrigation may allow agroecosystems to maintain non-drought levels of NEE, the associated energy inputs result in a move away from biophysical sustainability. In other words, more work is required to overcome gradients created by an irrigated system in drought conditions, and the system may remain "forced" in a negative feedback loop where it is not experiencing changes for adaptation at the ecosystem scale. One of the important theoretical underpinnings of this study is that evolution of spatial interactions and processes between abiotic and biotic factors involved in ecosystem development and agroecosystem production may influence the geographic variability of sustainability.

5.1.2 Chapter 3

According to our findings in Chapter 3, metrics of patch density and landscape shape index can be used with MODIS NDVI (to calculate fractional vegetation cover, Fr) and land surface temperature (LST) products at the 1-km scale to asses the sustainability of urban forms. Across scales from 30-m Landsat to 1-km MODIS scenes, patch density and landscape shape index provide reliable and predictable quantifications of urban density and urban complexity or mixed land uses. However, studies of sustainable urban form must also quantify the influence of forms on the mitigation of the urban heat island. Therefore, the availability of MODIS products, like NDVI and LST, at a high temporal resolution and broad spatial resolution are ideal for the assessment of biophysical sustainability at the global scale. By developing a conceptual framework based on the triangle method (Carlson & Arthur, 2000; Carlson, 2007; Gillies et al., 1997; Yuan & Bauer, 2007), we can compare pixels within cities, sections of cities, and different cities in LST-Fr space while considering the tradeoffs between density and complexity that add or take away from efforts for urban cooling.

In our case study of the São Paulo Metropolitan Region (SPMR), urban complexity, density and temperatures increase from 2004 to 2014, while urban greening decreases in most urban classes. From 2009-2014, policies associated with efforts for urban greening may be evident in the class with the least amount of vegetation or greatest amount of impervious surface area. Yet, this effort seems to have little if any effect on urban cooling.

5.1.3 Chapter 4

Throughout the Tiquié basin, studied in Chapter 4, high spatial and temporal variability in precipitation and river levels are likely to complicate climate change and sustainability analyses. In an initial attempt to address this complexity, Western researchers have attempted to bridge Indigenous knowledge (IK) and Western knowledge (WK) systems to provide climate services for sustainable livelihoods and orient future research. Using methodology that could be employed with other Indigenous or non-Indigenous astronomical and ecological calendars, timescales of relevance for evaluating and monitoring sustainability in agroecosystems and fisheries have been identified. Bridging the Tukano ecological calendar with classification trees and wavelet analysis, a recent trend in the increase of precipitation variance at the 4-day scale at Pirarara Poço is observed. This temporal scale is important to the slash-and-burn practice in Indigenous agroecosystems for bitter manioc production.

While the coexistence of IK and WK systems in this study identified specific timescales of relevance, it remains unclear whether this climate information will be useful in supporting collective continuance or Indigenous adaptive capacity and reducing climate change impacts and vulnerability. It is our hope that the methodology employed has resulted in knowledge that can be more easily incorporated into the Tukano cultural systems (Davis, 2006). Yet, "integration" of WK with IK will only occur if AIMAs and their communities decide the information from the Western climate analysis is useful for planning agricultural and fisheries activities.

5.2 Summary of Research Contributions and Policy Considerations

Both the thermodynamic measure of biophysical sustainability and the indicator of Indigenous ecological calendars in agroecosystems contribute to SDGs by promoting sustainable agriculture (Goal 2), combating climate change impacts (Goal 13), and protecting, restoring and promoting sustainable use of terrestrial ecosystems (Goal 15). The thermodynamic metric in Chapter 2 provides a pathway toward more sustainable agroecosystems during drought conditions, which protects and promotes the sustainability of terrestrial ecosystems by allowing crops to adapt and evolve based on spatial interactions that are not "forced." The pathway toward more sustainable agriculture points toward increased reliance on rainfed agroecosystems. Ironically, the study in Chapter 4 reveals a situation where increased rain frequency may be threatening the sustainability of Indigenous crop production. While Indigenous knowledge has been proposed as a new sustainability indicator for evaluating biodiversity loss (ICSU, 2015), the study in Chapter 4 also shows that IK can be used as an indicator for many other important sustainability goals including sustainable agriculture and combating climate change impacts.

The urban sustainability metrics and conceptual framework in Chapter 3 contribute to making cities resilient and sustainable (Goal 11) and to combating climate change and its impacts (Goal 13). Mitigating the urban heat island in cities is crucial to avoiding heat-related deaths and maintaining livable cities for future generations. Urban heat island mitigation is a long-term effort that requires an examination of tradeoffs and substantial investments in urban planning. The conceptual framework proposed in Chapter 3 with the use of global satellite data offers an inexpensive way of monitoring progress toward mitigation and biophysical sustainability.

Whether it's mitigating the urban heat island, decreasing irrigation use in drought conditions, or promoting the coexistence of Indigenous knowledge for climate services and adaptation, all of the studies in this dissertation provide pathways toward more sustainable livelihoods. These pathways go beyond identifying indicators and metrics that can help achieve biophysical targets; they also highlight policy considerations for drought management plans, urban planning, and national climate change policies.

Water rights are highly contested and can be the source of great conflict. Planning for drought-response measures is fraught with uncertainty and both long- and short-term considerations. While drought management plans around the world may outline actions for guaranteeing essential demands from reservoirs during drought conditions, these adaptation efforts are often not implemented early enough. Considering our findings from Chapter 2, policy-makers may want to limit irrigation earlier in drought conditions. Although such regulations may improve long-term agricultural sustainability, short-term financial and crop-loss insurance considerations may call for a system-wide review of current federal policies.

In addition, the water crisis that now faces states like California, US and São Paulo, Brazil leaves little room for expansion of irrigation practices. According to the Comprehensive Assessment of Water Management in Agriculture (Earthscan, 2007), there is room instead for upgrades in the rainfed agricultural sector. Our research findings support the investment in technologies, practices, and water governance for improving rainfed agriculture around the world in lieu of irrigated agriculture as highlighted by Rockström et al. (2010). Since our results also suggest that long-term GHG mitigation is more sustainable in rainfed agroecosystems, policies and practices promoting climate-smart agriculture should consider the tradeoffs between short- versus long-term carbon uptake by agriculture and how that relates to sustainability.

Although São Paulo participates in the Cities for Climate Protection (CCP) program and the

Energy Efficiency Program for the State of São Paulo, and has passed Municipal Law 14.933 in June 2009 to establish a target of 30% reduction in GHG emissions by 2012 (D'Almeida Martins & da Costa Ferreira, 2011), its urban heat island continues to grow. This situation exemplifies the ever-widening sustainability gap discussed by Fischer et al. (2007a), where to achieve sustainability we must: 1) identify meaningful biophysical targets, 2) apply policy tools to promote sustainable actions, and 3) identify key foundational or ethical values that link short-term policy actions with long-term sustainability targets. In Chapter 3, the use of biophysical metrics of LST and Fr combined with landscape metrics offers a clear conceptual framework for the identification of a meaningful biophysical target to reduce the urban heat island. This is the first step in identifying policy tools and actions that can reduce the sustainability gap.

Based on our findings at the 1-km scale and our knowledge of recent policy measures, we can make recommendations for future policy considerations at neighborhood and finer scales. In SPMR, measures calling for increased urban density in commercial and job centers to reduce commuting time should be reevaluated and contributions to the urban heat island should be quantified in relation to current urban densities. The distribution of increased urban greenspaces and trees planted throughout the city should be evaluated to maximize the cooling impact. In addition, implementation of policies requiring new construction as well as retrofitting measures in SPMR should now focus on thermal efficiency from energy use and building materials.

Following a presidential decree in 2009, Brazil adopted a National Plan on Climate Change (Law 12187/2009) in line with provisions of the UN Framework Convention on Climate Change. The law established a voluntary national reduction target of 36.1-38.9% of 2000-level emissions by 2020 (Law12187, 2009). In addition, it specified reduction targets for sectors including deforestation, agriculture, livestock, and energy. While the law required mitigation actions to be quantifiable for verification, no quantitative target for adaptation, losses and damage has yet been enacted by Brazilian climate change law. A national adaptation plan is currently being developed and has already defined a set of 10 priority areas for action including forests and biodiversity (personal communication with Melissa Oliveira).

Since Indigenous Territories constitute 13% of Brazil's total land area, Indigenous peoples and their knowledge should play a role in the creation of national policies on climate change adaptation and mitigation. Indigenous participation in monitoring and interpretation of climatic changes can be essential for distinguishing between socio-economic and climate change impacts. The research process in Chapter 4 illustrates how continued efforts to bridge Indigenous and Western knowledge systems can be managed to provide climate services and maintain sustainable environmental management in Indigenous territories where traditional management may be challenged in the context of global climate change.

Throughout the dissertation, ecological theory, knowledge production frameworks, and physical (thermodynamic) laws are examined and applied to contribute to an improved understanding of how sustainability interacts with climate change. The case study in Chapter 2 is a first attempt to unify theories of complex ecosystem dynamics with thermodynamic laws to better define sustainability. Based on the mechanisms involved in Wallace's evolutionary theory at the ecosystem scale, the study builds on theories of adaptation and resilience. By comparing thermodynamic entropy production with soil water content, energy partitioning, and the net ecosystem exchange of carbon, processes related to irrigation and carbon uptake that underly ecological considerations for sustainability in agroecosystems are recognized. The resulting pathway identified for sustainability in agroecosystems involves the tradeoffs between irrigation for climate change adaptation, food production, and greenhouse gas mitigation.

Findings from Chapter 2 further contribute to discussions concerning non-equilibrium dynamics, irreversible processes, and the role of thermodynamics in abiotic-biotic or spatial interactions (Brunsell et al., 2011; Holdaway et al., 2010; Kleidon, 2009; Perry, 2002; Phillips, 1999, 2008; Smith, 2005; Steinborn & Svirezhev, 2000; Svirezhev, 2000), which are applicable to all natural and anthropogenic landscapes on varying temporal and spatial scales. In addition, our methodology and analysis tying together thermodynamic laws, theories of complex ecosystem dynamics, and resilience theory in agricultural systems may be unique to sustainability science.

Throughout the application of the metrics and indicators, the identification of temporal and spatial scales has been a key factor in being able to disentangle the simultaneous ambiguity and complexity of sustainability definitions and assessments. Identifying appropriate temporal and spatial boundaries for assessments of human-environment systems continues to be a challenge in sustainability science. The use of satellite remote sensing data allows for scale-wise analyses of metrics like the research in Chapter 3. Knowing how metrics and indicators scale from building to neighborhood and landscape levels, for example, is crucial for setting, monitoring, and evaluating sustainability targets and creating effective policies. However, care must also be taken to consider cross-scale leakage and legacy impacts.

In addition, developing conceptual frameworks for sustainability that can be applied at multiple spatial and temporal scales is key to evaluating sustainability in light of global climate change. The use of readily-available MODIS satellite data with the conceptual framework in Chapter 3 provides an example of how urban sustainability assessments could be applied worldwide. The signal of rising global temperatures due to anthropogenic climate change will be automatically taken into account in these analyses, which links efforts toward mitigating urban heat islands at the local level to regional and global activities and broader accountability.

As shown in Chapter 4, potential impacts of global climate change are a justice issue for Indigenous peoples and other traditional peoples around the world who depend on land or riverine systems for their livelihoods. Efforts to bridge Indigenous and Western knowledge systems are vital for sustainable environmental management in Indigenous territories and other regions where traditional management and systems of responsibilities may be challenged in the context of global climate change (Berkes, 2006; Whyte, 2013). The effort in Chapter 4 contributes to theoretical and conceptual frameworks for bridging diverse knowledge systems by leveraging the work of boundary organizations, like the Socio-environmental Institute (ISA), and simultaneously addressing the coexistence of diverse knowledge systems and spatial and temporal scales of relevance for climate change adaptation (Berkes, 2006).

Therefore, another major contribution of this dissertation research is the value of cross-disciplinary research for sustainability science. Whether it is the multi-disciplinarity of Chapter 2 where theories from biogeography and ecology and physical laws are combined, the interdisciplinary research from Chapter 3 that calls on urban planning, landscape ecology, and remote sensing, or the transdisciplinary project in Chapter 4 that bridges Indigenous and Western knowledge systems, sustainability science necessitates access to a broad base of knowledge systems and disciplinary methodologies (Stock & Burton, 2011).

5.3 Recommendations for Future Work

Based on the research completed for this dissertation, three main strategies/guidelines for future work in sustainability science are recommended:

1. Evaluations of sustainability can be improved with inclusion of climate variables (or climate change projections)

2. Useful indicators and metrics are ones that help identify processes and mechanisms for pathways toward sustainability

3. Co-production of knowledge from various local knowledge systems and disciplinary methodologies will help identify appropriate spatial and temporal boundaries for sustainability assessments

Some of these suggestions are incorporated in my brief response to the second public consultation of the Report of the Open Working Group of the General Assembly on Sustainable Development Goals (UN A/68/970) in January 2015 found in Appendix A.

While tradeoffs and synergies between biophysical indicators of sustainability in agricultural, forest, and urban systems abound, the ultimate determinant in the sustainability arena is global climate change. The planetary boundaries referred to by Rockström et al. (2009) are eventually all impacted by this global phenomenon, and the foremost SDG should be to "Take urgent action to combat climate change and its impacts" (Goal 11). All economic and social systems depend on climate change mitigation and adaptation. And, the good news is that there are many win-win scenarios for addressing climate change and sustainability simultaneously. However, these actions will require considerable financial investments, lifestyle sacrifices, and institutional or policy reforms, along with continued scientific research.

Physical geographers possess the tools to advance theoretical frameworks, methodologies, and analyses to continue identifying processes and mechanisms that characterize the evolution of spatial interactions for sustainability. To further understand how thermodynamic metrics can contribute to pathways toward agroecosystem sustainability, researchers may be interested in evaluating how spatial interactions influence processes involved in native ecosystem development versus agroecosystem production. With available data from eddy-covariance (EC) towers, satellite remote sensing instruments, and reanalysis output, opportunities to empirically study these interactions and processes exist around the globe.

Figuring out how tradeoffs between urban complexity and density relate to LST-Fr space in different parts of the world will also be useful for defining more sustainable urban forms. With new urban classifications, like the Local Climate Zones (LCZs) scheme, quantification of urban heat islands will be more consistent across countries. Cities that have already been classified using LCZs provide opportunities for evaluations of the LST-Fr space in different bioclimatic zones and cultures.

In the Tiquié River basin, further research to uncover climate change and socio-economic impacts to precipitation at the 4-day scale is needed. Wavelet coherence transforms can be used to evaluate possible teleconnections with sea surface temperatures in the Tropical Atlantic, the Madden-Julian Oscillation (MJO), the El Niño Southern Oscillation (ENSO), and the Pacific Decadal Oscillation (PDO). Essential to the co-production of knowledge in this region is the dissemination of results to the communities of the Tiquié and continued Indigenous participation in interpretation of results for distinguishing between socio-economic and climate change impacts. In addition, the methodology in Chapter 4 can be expanded to other Indigenous territories to uncover spatial and temporal variability of flood and drought patterns across the Amazon basin to help build adaptive capacity and support emerging systems of responsibilities in Indigenous communities.

The studies in this dissertation are merely initial steps taken to consider biophysical metrics and indicators of sustainability without direct consideration of social and economic indicators. Evaluating biophysical sustainability alone does not, however, guarantee that cultural, social, and economic components of a system cannot compensate for temporary biophysical failures. For this reason, the addition of social and economic factors to complement these case studies is necessary for an overall assessment of sustainability.

Definitions of sustainability in agricultural and urban systems may still require adjustments, but grounding definitions with specific indicators and metrics is essential to working toward more sustainable systems. When we consider that, due to human-caused climate change, our Earth system is now working in a "no-analogue state," defining sustainability actually becomes easier. The creation of the Sustainable Development Goals offers a great opportunity to look at sustainability from a different perspective. Based on scientific theories and laws, and with the right methodology and tools, spatial and temporal boundaries and useful indicators or metrics can be identified to compare biophysical systems not just as they were historically but as they can be or need to be to support human life.

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Appendix A

Response to Second Public Consultation for Sustainable Development Goals

The comments below offer a brief response to the request for feedback on Indicators and Monitoring Framework for SDGs.

1) Need for a Weighted Framework. The report mentions a "universal, holistic framework" to address the "dimensions of economic development, social inclusion, and environmental sustainability." I agree with Griggs et al. (2013) that we need to move beyond creating SDGs based on the triple, parallel pillars of sustainability. Instead, we need to consider a hierarchical framework, where ecological or environmental considerations are first (Fischer et al., 2007). Therefore, the monitoring framework outlined should be a weighted system that gives priority to biophysical or environmental indicators first, such as those related to climate change.

The separation of the monitoring framework into spatial (global, national, regional) and temporal (annual reporting) boundaries is helpful. However, I wonder how leakage and legacy impacts will be ascribed.

2) Fewer Goals. The goals can be condensed as follows:

1. Goal 3: Ensure healthy lives and promote well-being for all at all ages —includes Goals 1, 2, 6, 11

2. Goal 4: Ensure inclusive and equitable quality education and promote life-long learning opportunities for all —includes Goal 5

3. Goal 7: Ensure access to affordable, reliable, sustainable, and modern energy for all —includes Goal 9

4. Goal 10: Reduce inequality within and among countries —includes all Goals, especially Goals 1, 2, 4, 8, 9, 11, 16, 17

5. Goal 12: Ensure sustainable consumption and production patterns —includes Goals 14, 15, 17

6. Goal 13: Take urgent action to combat climate change and its impacts —includes all Goals, but particularly Goals 9, 14, 15, 17

3) Useful Indicators. The links between indicators and sources of available data through the World Bank, UN, FAO, IEA, etc. should be helpful to nations. In addition, the incorporation of satellite remote sensing data can provide real-time evaluations. However, we need to make sure that all indicators relate directly to specified environmental, social or economic processes or mechanisms, so they can be altered to work toward greater sustainability. Otherwise, an indicator may be useless. For example, there is a big mechanistic gap between monitoring poverty levels and reaching a goal of ending poverty. How will we make sure that the mechanisms to address certain indicators and reach certain goals are sustainable in themselves?